BALTIC SEA ENVIRONMENT PROCEEDINGS

No. 35 B

SECOND PERIODIC ASSESSMENT OF THE STATE OF THE MARINE ENVIRONMENT OF THE BALTIC SEA, 1984 – 1988; BACKGROUND DOCUMENT

BALTIC MARINE ENVIRONMENT PROTECTION COMMISSION
– HELSINKI COMMISSION –

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P R E F A C E

Within the framework of the Baltic Marine Environment Protection Commission - Helsinki Commission - marine environment monitoring data have been collected since 1979 within the frame of the Baltic Monitoring Programme (BMP). The guidelines for the programme are reviewed every five years by the Commission and the revised guidelines are published in the Baltic Sea Environment Proceedings (BSEP) series. The Third Stage of the Baltic Monitoring Programme (BMP) started in 1989 and the Guidelines were published in Baltic Sea Environment Proceedings Nos. 27A, B, C, D. The monitoring data provided by all Baltic Sea States are stored and processed in the HELCOM Data Base established by the Commission on a consultant basis. The aim of the common data bank is to serve as a source of current information on the state of the Baltic Sea. The periodic assessments of the state of the marine environment of the Baltic Sea are published by the Commission regularly as a comprehensive scientific overview and as general conclusions drawn on the basis of this scientific overview. The previous assessments were published in the Baltic Sea Environment Proceedings Nos. 5A and 5B (1981) and Nos. 17A and 17B (1986/1987). The conclusions drawn from the present scientific evaluation have been published by the Commission in the Baltic Sea Environment Proceedings No. 35A (Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1984-1988; General Conclusions) and presented to the Baltic Sea Conference, held in Ronneby, Sweden, 2-3 September 1990.

The present publication, BSEP 35 B, contains the chapterwise scientific evaluation for the years 1984-1988. The work has been done by the ad hoc Group of Experts for the Preparation of the Second Periodic Assessment of the State of the Baltic Sea (GESPA), established by the Helsinki Commission. Experts from all Baltic Sea States participated in the work of the group, as well as representatives of the International Council for the Exploration of the Sea (ICES), the Baltic Marine Biologists (BMB) and the Conferences of the Baltic Oceanographers (CBO). Professor Sebastian A. Gerlach from the Federal Republic of Germany acted as Chairman, and the Environment Secretary of the Commission, Ms. Terttu Melvasalo, as Secretary of the Group. Each chapter in this proceedings presents results emerging from the monitoring activities, other data available, scientific literature and other relevant information, evaluated by experts and coordinated by the Convener and Co-convener of each chapter, and written by individual authors.

It is my sad duty to recall that Professor Aarno Voipio from Finland, one of the well-known experts of the project and the first Executive Secretary of the Helsinki Commission, died during the final stage of the completion of this project, 11 February 1990.

On behalf of the Helsinki Commission, sincere gratitude is expressed to the editor, Professor Sebastian A. Gerlach, and to all scientists who assisted in carrying out the project, to Mr. Klaus Reiber of the Marine Research Institute of the University of Kiel, who assisted in redrawing most of the figures, to Ms. Teija-Liisa Lehtinen from the Helsinki Commission and Ms. Pia Kostakow from Finland, who were responsible for typing and technical editing of the documents, as well as to the Federal Maritime and Hydrographic Agency in Hamburg which has undertaken the printing of both volumes of Baltic Sea Environment Proceedings No.35.

Helsinki, 15 October 1990

Fleming Otzen
Executive Secretary
Helsinki Commission

Baltic Sea Environment Proceedings 35B (1990)
Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1984-1988 ; Background Document

INTRODUCTION

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Scientists from the seven countries bordering the Baltic Sea worked together in the "Ad hoc Group of Experts for the Preparation of the Second Periodic Assessment" (GESPA) of the Helsinki Commission and compiled data from the monitoring period 1984-1988. They evaluated the actual levels and concentrations measured and indicated trends in comparison with the previous assessment period 1979-1983 and with older data, referring to meteorology and hydrography, oxygen conditions and nutrients concentrations, phytoplankton, zooplankton, zoobenthos, fish and micro-organisms, and heavy metals and organic contaminants in sea water and biota. Results and conclusions are presented in the respective chapters of the volume. The executive summary contained in the Baltic Sea Environment Proceedings No. 35A (1990) "Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1984-1988; General Conclusions", agreed by the conveners, co-conveners and experts involved in the preparation of the Second Periodic Assessment at their last meeting in April 1990, is contained in this introduction under paragraph 5.

1. The Baseline Assessment 1980

In 1978 the Interim Helsinki Commission asked for an assessment of existing data on the pollution of the Baltic Sea. Under the editorship of Terttu Melvasalo who was assisted by co-editors Janet Pawlak, Klaus Grasshoff, Lars Thorell and Alla Tsiban, about 30 experts from the seven Baltic countries worked out the "Assessment of the Effects of Pollution on the Natural Resources of the Baltic Sea, 1980 (Melvasalo et al., 1981).

Topics treated in chapters were
- Physical parameters
- Dissolved gases
- Nutrients
- Harmful substances
- Biological parameters.

Information was ***given on***
- Relevance of methods
- Gaps in knowledge
- Trends
- Differences in sub-areas
- Inter-relationships with other processes
- Inputs to the Baltic Sea
- Effect of human activities
- Degree of pollution.

By "assessment" the Helsinki Commission meant an evaluation or judgement of the conditions and quality of the environment and its living organisms. The 1980 assessment was meant as a "baseline assessment" from where future trends should be determined in subsequent assessments which could make use of data from the Baltic Monitoring Programme. The 1980 assessment, including a bathymetric chart of the Baltic Sea, is still a valuable overview of the conditions in the Baltic. In the same year the book "The Baltic Sea" by A. Voipio (1981) appeared with additional comprehensive information.

2. The First Periodic Assessment for the period 1979-1983

Guidelines for the first stage of the Baltic Monitoring Programme were issued on August 25, 1980. However, already in 1979 the Baltic Monitoring Programme started, and not later than 1981 the Helsinki Commission decided to establish the "Ad hoc Group of Experts on Assessment of the State of the Marine Environment of the Baltic Sea" (GEA). More than 40 experts from all seven countries bordering the Baltic Sea joined in a cooperative effort and produced, under the chairmanship of Julius Lassig, the "First Periodic Assessment of the State of the Marine Environment of the Baltic Sea Area, 1980-1985". The general conclusions appeared in 1986, the background document in 1987 (Baltic Marine Environment Protection Commission 1986; 1987 a).

Chapters, reflecting the sub-divisions of the guidelines, were

Hydrography
Nutrients
Harmful substances
Pelagic biology
Zoobenthos
Microbiology.

The GEA evaluated the results of the Baltic Monitoring Programme for the period about 1979 to 1983 and reviewed other relevant information pertinent to the assessment of the state of the Baltic Sea. In writing the respective chapters, the conveners and experts of the sub-groups had considerable freedom. Chapters of the background document were issued under the names of those scientists who wrote the chapters. In spite of many shortcomings in the initial phase of the Baltic Monitoring Programme, the First Periodic Assessment is a unique example how monitoring should be done: it must be followed by an assessment of the monitoring data organized as rigid as the monitoring itself. The results of the assessment are meant to improve the quality of the next monitoring period, they point to deficiencies of knowledge and summarize what can be said about the environmental trends, referring not only to the assessed five-year period but considering the past in general. Finally, proposals for action required were included in the assessment report.

3. General conclusions from the First Periodic Assessment, and from other sources, regarding the state of the Baltic Sea in the period 1979-1983

In the assessment period 1979-1983 concentrations of cadmium, copper and nickel in offshore Baltic Sea waters were higher than in the waters of the open North Atlantic. This has partly anthropogenic causes, partly it can be explained, in the Baltic Sea, by the high percentage of river water which by nature has higher concentrations of some heavy metals compared with ocean water. Concentrations of mercury and lead in offshore Baltic Sea waters were in the same range as in the open North Atlantic. Mercury concentrations in fish from the offshore Baltic were likewise not significantly higher than in fish from the North Atlantic. In 1983 there was no mercury problem evident in the offshore Baltic, in spite of the fact that there were still local hot spots in coastal areas. Concentrations of DDT (and metabolites) in eggs of sea birds and in herring from the Baltic Sea were in 1979-1983 still higher compared with data from the North Sea, however, there had been a ten-fold reduction of concentrations achieved between 1970 and 1980. With regard to mercury and DDT, by 1979 environmental protection measures have been effective. PCB concentrations, however, seemed to be still rather high in 1983. Very little was known regarding other organic contaminants.

Oxygen is transported to the deep basins of the Baltic Sea with episodic salt water inflows which occur during autumn to spring. The last major inflow to the Baltic Proper was observed at the Darss Sill in autumn 1976. Following this oxygen supply, many previously anoxic deep areas were recolonised by some macrofauna. However, during 1979-1983 the deeps in the Central and Northern Baltic Proper did not get new supplies of oxygen. The deep water was stagnant, anoxic, and by bacterial metabolism hydrogen sulphide developed and killed all zoobenthos. A small 1979/1980 inflow permitted macrofauna populations to increase temporarily only in the deeps of the Southern Baltic Proper. By 1983, the oxygen conditions had again deteriorated to the poor situation prevailing before the 1975/1976 inflow. 1981 was the turning point to bad late summer oxygen situations in the Southern Kattegat, in Little Belt, in Kiel Bay and in the Bay of Mecklenburg, documented by the die-off of macrozoobenthos in many sub-pycnocline waters.

Unfortunately, oceanographers cannot measure well enough the water transport through the Sound and through Great Belt and Little Belt. There are methodological problems with current measurements in the complicated system characterised by complex water movements back and forth and by successive entrainment of ingoing water into the outgoing water mass. Therefore, oceanographers in 1986 could not make good calculations on the year to year transport of water from the Skagerrak to the Baltic Sea, neither during episodic inflow events in winter nor under stratified summer situations, when the inflow is driven by the estuarine circulation (Falkenmark 1986). Modelling the water transport in and out the Baltic Sea is still in its infancy.

It is evident that large amounts of all kinds of harmful substances and plant nutrients, are introduced to the Baltic Sea via rivers. In fact, the Baltic Sea is just a continuation of the many rivers which drain 1,670,00 km^2, an area four times larger than the Baltic Sea itself

(390,000 km^2 without the Kattegat; Fig. 1, Table 1). The seven major rivers together (Table 2) contributed in 1981 with 211 km^3 about half of the 439 km^3 of freshwater which was the mean annual discharge in the period 1951 to 1970 to the Baltic Sea (Kattegat excluded, Table 1).

Figure 1. Sub-regions and catchment area (drainage basin) of the Baltic Sea. Catchment area from Mikulski (1986), sub-regions according to Baltic Marine Environment Protection Commission (1988):

A Bothnian Bay
B Quark
C Bothnian Sea
D Åland Sea
E Archipelago Sea
F Gulf of Finland
G Gulf of Riga
H Northern Baltic Proper
I Western Gotland Basin
J Eastern Gotland Basin
K Southern Baltic Proper
L Bay of Gdansk
M Bay of Mecklenburg
N Kiel Bay
0 Little Belt
P Great Belt
Q Sound
R Kattegat
S Skagerrak

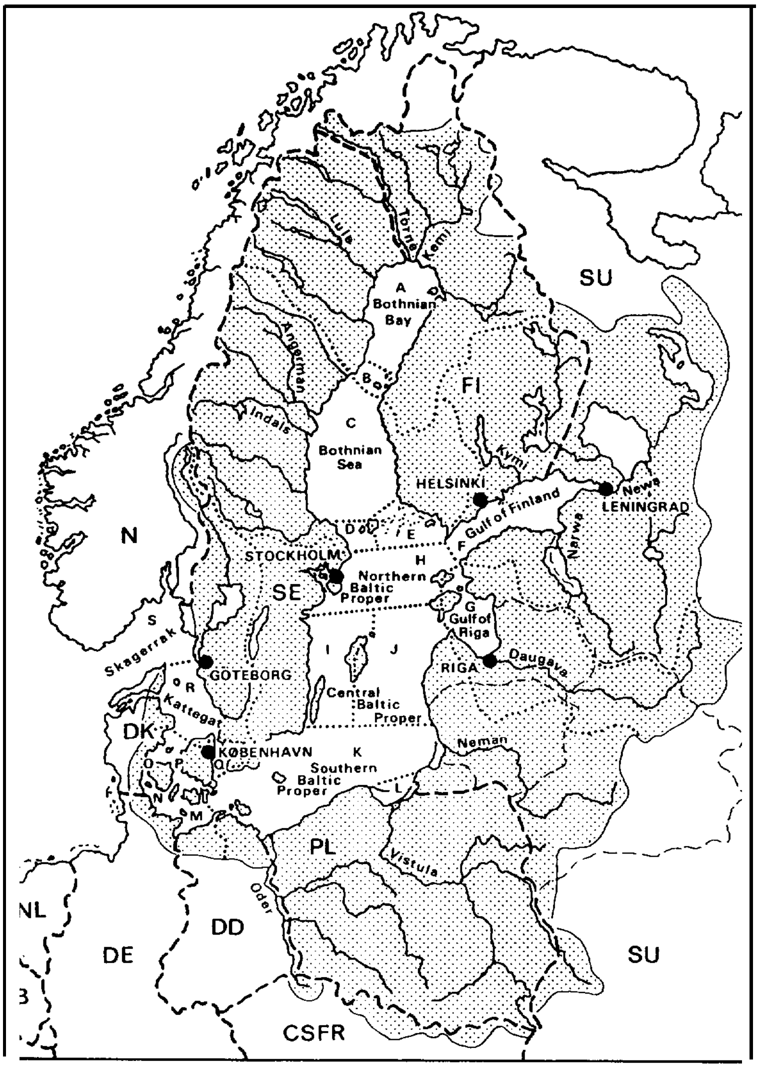


Table 1. Geographical data for the Baltic Sea. Areas, volumes and river flow (mean of period 1951-1970) from Mikulski (1986; partly modified in conformity with data used for the preparation of the Second Pollution Load Compilation of HELCOM); these corrections and data on catchment area (= area of drainage basin) and population (number of persons living in the catchment area) by courtesy of Mr. R. Rudloff, Deutsches Hydrographisches Institut, Hamburg. According to Rybinski et al. (1989), about 970 000 additional inhabitants of Czechoslovakia drain into the Baltic Proper.

Sub-region	Sea area km^2	Volume km^3	Country	Catchment area km^2	Flow km^3/y	Population
Bothnian Bay	36 260	1 500	FI	146 000	50	241 000
(incl. Quark)			SE	131 000	55	472 000
Bothnian Sea	79 256	4 889	FI	39 000	11	430 000
(incl. Åland Sea)			SE	180 000	74	1 045 000
Archipelago Sea	incl.	incl.	FI	9 000	2.6	240 000
Total Gulf of Bothnia	115 517	about 6 370		505 000	193	2 428 000
Gulf of Finland	29 600	1 100	FI	50 000	11	1 961 000
			SU	374 000	100	10 000 000
Gulf of Riga	13 839	408	SU	117 550	27	4 000 000
Total Gulfs of Finland and Riga	43 439	1 508		541 550	138	16 000 000
Baltic Proper			SU	114 820	31	5 500 000
			PL	312 000	51	32 800 000
			SE	72 000	17	4 244 000
			DK	1 068		78 000
			DD	15 980		906 000
Total	211 096	13 045		515 868	100	44 000 000
Belts and Western Bays	18 273	262	DD	10 191		1 060 000
			DE	5 450		1 100 000
			DK	10 000		1 600 000
Sound	1 848	25	DK	1 900		1 682 000
			SE	2 500		724 000
Total Sound and Belts	20 121	287		30 041	8	6 166 000
Kattegat			DK	7 000		465 000
			SE	70 000		2 200 000
Total	22 387	421		77 000	29	2 665 000
Total Baltic Sea	412 560	21 631		1 669 459	468	71 000 000

Table 2. Fresh water discharge 1981 of the seven major Baltic rivers. From United Nations Statistical Commission (1987).

River	Catchment area km²	River Flow m³/s	River Flow km³/year
Neva	281 000	2 463	77.7
Vistula (Wisla)	194 424	1 040	32.8
Kemi (Kemijoki)	51 400	739	23.3
Daugava	87 900	688	21.7
Neman	92 200	674	21.3
Oder (Odra)	118 861	560	17.7
Kymi (Kymijoki)	37 235	517	16.3

The amount of freshwater delivered to the Baltic Sea from rivers varies from year to year between about 420 and 550 km^3/year. This variation together with the salt water inflows influence salinity and stratification of the water masses of the Baltic Sea. Salinity increased in the seventies, but has been decreasing since then.

Total loads of nutrients and other substances from land and from the atmosphere to the Baltic Sea have been estimated by the Helsinki Commission according to data which are not strictly comparable and are partly incomplete so that the First Pollution Load Compilation must be interpreted with great care (Baltic Marine Environment Protection Commission 1987 b). Estimated annual inputs to the Baltic Sea are listed in Table 3.

Table 3. Estimates of the total pollution load into the Baltic Sea, including the Kattegat. Input from land: from Baltic Marine Environment Protection Commission (1987 b). Input 1986 from atmosphere: from Areskoug 1989.

Contaminant	Input from land t/year	Atmospheric input t/year
Degradeable organic matter (BOD equivalent)	1 640 000	
Total phosphorus	48 500	
Total nitrogen	530 000	270-630 000
Cadmium	59	35
Copper	4 200	470
Lead	?	1 560
Zinc	8 900	3 400

Larsson et al. (1985) have estimated an eight-fold increase of phosphorus loads to the Baltic Sea since the beginning of the century, a four-fold increase since 1950. The increase of nitrogen loads between 1950 and 1980 was probably by two.

Between 1970 and 1980 phosphate and nitrate concentrations increased three-fold in the surface winter water of the Baltic Proper. There was an increase of salinity, too, indicating increased mixing with deep water which is rich in salt and in nutrients. This increased mixing was partly responsible for the nutrient increase. However, it is reasonable to conclude that anthropogenic nutrient inputs made a considerable contribution and therefore enhanced the process of primary production in the surface layer. Hence anthropogenic inputs are responsible for increased oxygen consumption in the Baltic deep water. This conclusion is not contradictory to the fact that the actual cause of oxygen depletion in the deep water is insufficient oxygen import, the lack of inflows of salt-rich water from the Skagerrak.

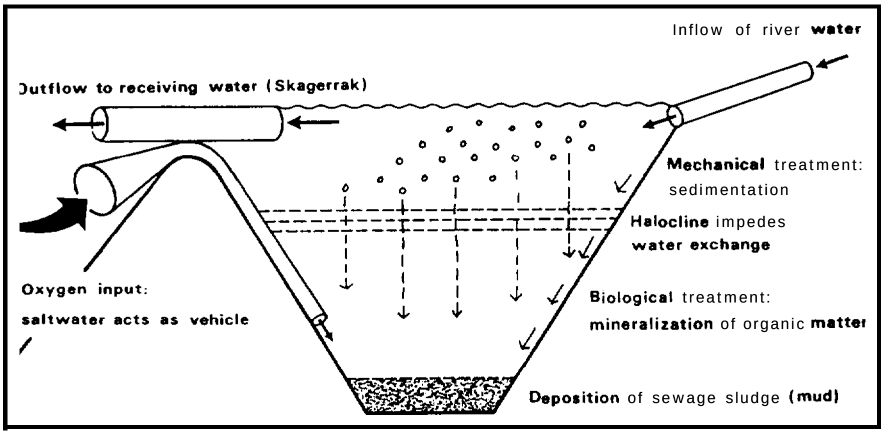
The main finding from the First Periodic Assessment was ongoing eutrophication in the Baltic Sea. This called for further action to reduce the inputs of nutrients from the Baltic Sea states (Baltic Marine Environment Protection Commission 1986, p. 13).

Nutrient concentrations increased in the 1970s in spite of the fact that Sweden and Finland were investing large sums of money into wastewater treatment plants which not only reduce the organic load and hence the oxygen demand of the wastewater, but are also equipped with phosphorus removal techniques. However, the bulk of nutrients was introduced into the Baltic Sea from the countries at the eastern shores. Therefore, the reduction of nutrient inputs in Denmark, Sweden and Finland had no apparent effect upon nutrient concentrations in offshore Baltic Sea water. However, the quality of coastal waters was improved.

There is a reason to believe that phytoplankton biomass and primary production in the Baltic Sea doubled between 1970 and 1980. In comparison with previous periods more food became available for secondary producers, so that more macrofauna biomass could develop in the sediments of shallow areas which were not disturbed by oxygen deficiency.

Inputs from land go into river mouths, into shallow lagoons and into coastal waters and are partly deposited there in the sediment, and partly transported into the offshore Baltic Sea. Many pollutants and nutrients have an affinity to particles: they are physico-chemically bound to clay, adhere to particle surfaces or are incorporated into living organisms, by affinity to lipids or by biochemical processes. The offshore Baltic Sea works in the same way as a wastewater treatment plant (Fig. 2), but at much lower concentrations: dissolved compounds are incorporated into particles which tend to sink to the sediment. By this process the water is cleaned. Contaminant concentrations in offshore Baltic water are surprisingly low, in spite of the many inputs of pollutants and nutrients. But one cannot clean something without getting something else filthy (one of "Murphy's Laws"). Pollutants and nutrients introduced to the Baltic Sea are eliminated via sedimentation from the water, but accumulate in the deep sediments, and are concentrated in the deepest parts like at the bottom of a funnel. A deposit of persistent toxic substances accumulates in the deep mud of the Baltic Sea. One cannot tolerate this toxic dumpsite in the long run. One should have in mind that mankind as well as fellow creatures, want to live decently for thousand and more years in and around the Baltic Sea. Inputs of toxic substances should be avoided.

Figure 2. The open Baltic Sea compared to the function of a wastewater treatment plant. After Gerlach (1988)



4. The Second Periodic Assessment for the period 1984-1988

As early as 1987 the Helsinki Commission asked for the Second Periodic Assessment which nominally should cover the period 1985 to 1990. The "Ad hoc Group of Experts for the Preparation of the Second Periodic Assessment" (GESPA), was established by the governments of the seven countries bordering the Baltic. The terms of reference were similar to those of the "Ad hoc Group of Experts on Assessment of the State of the Marine Environment of the Baltic Sea" (GEA). The sequence of chapters is different. The responsibility for sub-chapters is again on individual scientists.

The Group of Experts for the Second Periodic Assessment was established in 1987. GESPA 1 meeting was held in Kiel, 25-28 August 1987, followed by a meeting of conveners and co-conveners on board of "Georg Ots" and in Helsinki, 13-14 June 1988. GESPA 2 meeting was held again in Kiel, 6-9 September 1988, and GESPA 3 in Tallinn, 3-6 May 1989. Finally, there was a meeting of conveners and co-conveners 29-31 March 1990 and GESPA 4 meeting 2-5 April 1990 in Sopot for drafting the final reports.

5. Executive Summary of the General Conclusions of the Second Periodic Assessment (from Baltic Sea Environment Proceedings No. 35A)

The present assessment of the Baltic Sea Area concerning primarily the years 1984-1988 deals mainly with observations made in the open Baltic Sea and, consequently, the statements do not reflect findings in coastal areas, which will be assessed separately. In addition, a specific assessment on the state of Baltic sediments is in the final stage of preparation under the International Council for the Exploration of the Sea (ICES), which also prepares assessments on the state of Baltic seals on a regular basis. Furthermore, the document only occasionally covers information on the health of fish, birds and marine mammals.

Due to the ban in the use of some harmful substances, positive changes were observed; DDT and PCB concentrations in biota have decreased since the 1970s and are now on a lower and steady level, although comparable data on herring indicate that the levels are still higher in the Baltic than in the Skagerrak area. After the ban on technical hexachlorocyclohexane (HCH), the decrease of a-HCH concentrations in water is still continuing. Concentrations of organochlorine residues in fish from the Baltic Proper are still 3 to 10 times higher than in fish from around the Shetland Islands. Among the "new contaminants", there has been an increasing number of organic substances identified which are potentially harmful to the environment.

Trace element concentrations in fish and shellfish have not changed remarkably since the early 1980s. Generally, it can be stated that mercury concentrations in biota do not significantly differ now from those in the North Se;. and the North-East Atlantic. Compared with actual background levels, elevated mercury concentrations were only found in the Sound and in the southern Bothnian Sea. For the latter area, however, a considerable decrease of the concentrations could be identified during recent years.

A still upward trend of cadmium concentrations was observed in fish from the northern part of the Bothnian Bay. The reason for this is not fully understood. Other elements, such as zinc and copper, showed similar trends.

Fish and shellfish from sampling locations in the Kattegat and the Belt Sea showed tendencies for decreasing lead concentrations. It is possible that this is already an effect of the increased use of unleaded gasoline.

Meteorological conditions during the period 1984-1988 are characterised as variable: three unusually cold winters (1984/85 to 1986/87) with heavy ice conditions followed by two warm ones. The river runoff to the Baltic Sea was, in general, higher than the long-term mean, except the years 1985 and 1986.

Salinity continued to decrease mainly due to lack of major inflows of highly saline water from the North Sea during the last 13 years. Furthermore, temperature and density have decreased in the deeper layers of the Baltic Proper. The current stagnation period in the Eastern Gotland Basin is regarded as one of the longest and most serious stagnation intervals recorded during this century. This has caused the most extreme changes in the deep layers that have been observed since the beginning of oceanographic observations in the Baltic Sea.

The area with insufficient oxygen conditions for macrofauna (about 70 000 km^2 with less than 2ml/l oxygen in bottom water) has fluctuated in extent from year to year, but has not increased for 25 years in the Central Baltic Sea and the Gulf of Finland. However, due to the long stagnation period, the oxygen concentrations in the deeps of the Baltic Sea have continuously decreased and hydrogen sulphide concentrations in the deepest areas of the Eastern Gotland Basin are now the highest ever measured. Due to decreasing salinity and consequent lowering of the halocline associated with increased vertical exchange, oxygen has penetrated more deeply into the intermediate layers, at about 90-100 m in some areas, and has improved life conditions at the sea floor in this depth range.

In many areas of the Baltic Sea, the strong increase of phosphorus and nitrogen concentrations, which was observed in the 1970s, has stopped, with the exception of the Kattegat and the Gulf of Riga. Phosphorus and nitrogen concentrations, although no longer increasing in all parts of the Baltic Sea area since 1978, have recently been at such a high level that the increasing biological production and its subsequent sedimentation, followed by the microbial destruction of the biogenic organic material, cause further deterioration of the oxygen conditions in Baltic deep water. The high phosphate accumulation rates identified in the near-bottom water layers of the Central Baltic deeps since 1977 mainly result from phosphate remobilisation from the sediments due to the increasing hydrogen sulphide concentrations. Silicate concentrations have recently been decreasing, on average, in the surface layer of the Baltic Sea area.

Unusual algal blooms appear to occur more frequently in the Kattegat and the Belt Sea. There is evidence that phytoplankton primary production has doubled within the last 25 years in the area from the Kattegat to the Baltic Proper, with a similar doubling of phytoplankton biomass and its subsequent sedimentation. In the 1980s phytoplankton was at a high level, fluctuating from year to year according to the weather. The decomposition in the benthic system decreases oxygen levels in bottom waters. Consequently, low oxygen concentrations during late summer and autumn have often been observed in the southern Kattegat, the Belt Sea, the Öresund and the Arkona Sea in the eighties.

The increased frequency of poor oxygen conditions in the deep water has had a serious impact on the zoobenthos in the area from the southern Kattegat to the Arkona Basin, and on demersal fish and Norway lobster, pushing northwards into the Kattegat the southern boundary for commercial fishery of Norway lobster.

6. Events 1984-1988

In the period from 1984 to 1988 new methods were developed and introduced. For example the analysis of individual PCB congeners was developed, which means that previous PCB data are outdated. In the period from 1984 to 1988 new gaps of knowledge became evident, for example regarding the effects of nanogram per liter concentrations of tributyltin or the effects of chlorinated dioxins and furans in the picogram per liter range. For the first time, in the Kattegat the mass decease of common seals and an early summer bloom of the toxic alga *Chrysochromulina polylepis were* observed.

Governments of the Baltic Sea states have agreed to reduce inputs of pollutants by 50 %, by 1995, as stated in the Ministerial Declaration of the year 1988 (Baltic Marine Environment Protection Commission 1988 b).

7. Concluding remarks

In the 1970s, the salinity, and the concentrations of phosphorus and nitrogen of the Baltic Sea surface water continued to increase. There is evidence that also phytoplankton biomass and productivity increased during that period. Episodic inflows of high salinity water from the Kattegat renewed the oxygen content in the bottom water in the Baltic deeps. Several times macrofauna recolonisation took place in many of the deep areas. DDT-concentrations and mercury concentrations in fish and birds started to decrease as a consequence of reducing measures. At that time there was a lot to report, some news positive, others negative, for example that the Baltic Sea became increasingly anoxic in the deep. The exciting headline for this period of the 1970s could have been: there is increasing eutrophication due to the anthropogenic inputs of organic matter and nutrients.

During the period of the Second Periodic Assessment, 1984-1988, the changes were less marked. Since 197711978 salinity has been gradually decreasing because there have been no major inflows of high salinity water from the Kattegat. Consequently, since 10 years there has been no oxygen in the Gotland Deep. The exciting headline for the Second Periodic Assessment could be: the situation in the Baltic remained rather unchanged, except salinity, in spite of the fact that the monitoring period 1984 to 1988 was no average: three winters were abnormally cold, and two abnormally warm.

In the 1980s nutrient concentrations were at a high level and caused the phytoplankton to flourish, but nutrient concentrations did not further increase. This is not a proof that anthropogenic inputs did not further increase but only a signal that hydrographic conditions play an important role. It is open for speculations what will happen with eutrophication when major salt water inflows occur in the near future. As regards river transported heavy metals and organic contaminants, a higher fresh water proportion in the water of the Baltic Sea means higher concentrations compared to a situation when water in the Baltic Sea has higher salinity.

In a complicated way anthropogenic inputs and hydrographical processes work together resulting in the actual concentrations of plant nutrients and toxic substances.

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Appendix: The sequence of sub-regions in the presentation of the Second Periodic Assessment

The sequence of the Baltic Sea sub-regions followed in the Second Periodic Assessment is the same as in the First Periodic Assessment, starting with the Kattegat and ending up in the Gulf of Bothnia. In 1988 the Helsinki Commission developed an opposite sequence following the way of freshwater from the Gulfs to the Skagerrak, and introduced a new coding of the monitoring stations. The new arrangement is part of the Guidelines for the Baltic Monitoring Programme for the period 1989-1994 (Baltic Marine Environment Protection Commission 1988 a). In Table 4 the old and the new classifications are compared, for better understanding of the contents of the Second Periodic Assessment, which has to assess the past (and therefore makes use of the old coding of stations) but is also meant as a guide for the future. The location of the stations (with old and new codes) is depicted in Figs. 3-6. In Fig. 1 the new sub-regions of the Baltic Sea area are given, while Fig. 7 presents the traditional sub-regions with the names used in the Second Periodic Assessment.

Table 4. Baltic Sea sub-regions and monitoring stations with codes as used in the Second Periodic Assessment. New codes in brackets. After Baltic Marine Environment Protection Commission (1988).

A. The Transition Area		
Kattegat		
GF 4	76-80m	(= R 7)
Fladen	75-78m	(= R 6)
403	42-44m	(= R 5)
409	13-15m	(= R 4)
413	53-57m	(= R 3)
921	23-25m	(= R 2)
925	41-45m	(= R 1)
Sound		
431	48-51m	(= Q 2)
31 s	17-18m	(= Q 1)
Great Belt		
(no stations in Little Belt)		
935	46-50m	(= P 2)
939	36-40m	(= P 1)
Kiel Bay		
(including Fehmarnbelt)		
450	29-34m	(= N 4)
Kieler Bucht	18m	(= N 3)
Siiderfahrt	22m	(= N 2)
952 (Fehmarnbelt)	25-29m	(= N 1)
Bay of Mecklenburg		
Mecklenburger Bucht	23-27m	(= M 2)
954	22-24m	(= M 1)
B. The Baltic Proper		
Southern Baltic Proper		
Arkona Basin		
GDR 30	21m	(= K 8)
BY 1	44-46m	(= K 7)
441	23-25m	(= K 6)
GDR 113	48m	(= K 5)
BY 2	46-48m	(= K 4)
P 38		(= K 3)
Bornholm Basin		
BY 5	87-93m	(= K 2)
Bay of Gdansk		
P 1	110m	(= L 1)
Central Baltic Proper		
Eastern Gotland Basin		
BCS III 10	88-91m	(= K 1)
new station	47m	(= J 2)
BY 15	220-249m	(= J 1)
Western Gotland Basin		
BY 38	106-109m	(= I 1)
Northern Baltic Proper		
BY 31	420-450m	(= H 3)
BY 28	160-170m	(= H 2)
LL 12	81-84m	(= H 1)
C. The Gulfs		
Gulf of Finland		
(no stations in Gulf of Riga)		
LL 11	66-69m	(= F 5)
LL 11a	59m	(= F 4)
LL 7	80-100m	(= F 3)
LL 4a	58-64m	(= F 2)
LL 3a	64-66m	(= F 1)
Gulf of Bothnia		
Åland Sea		
(no stations in Archipelago Sea)		
F 64	280-293m	(= D 1)
Bothnian Sea		
SR 5	119-125m	(= C 4)
B VII	49-57m	(= C 3)
US 6b	77-83m	(= C 2)
US 5b	208-220m	(= C 1)
Bothnian Bay		
(no stations in Quark)		
Bo 3	103-108m	(= A 3)
c VI	68-72m	(= A 2)
F 2	85-91m	(= A 1)

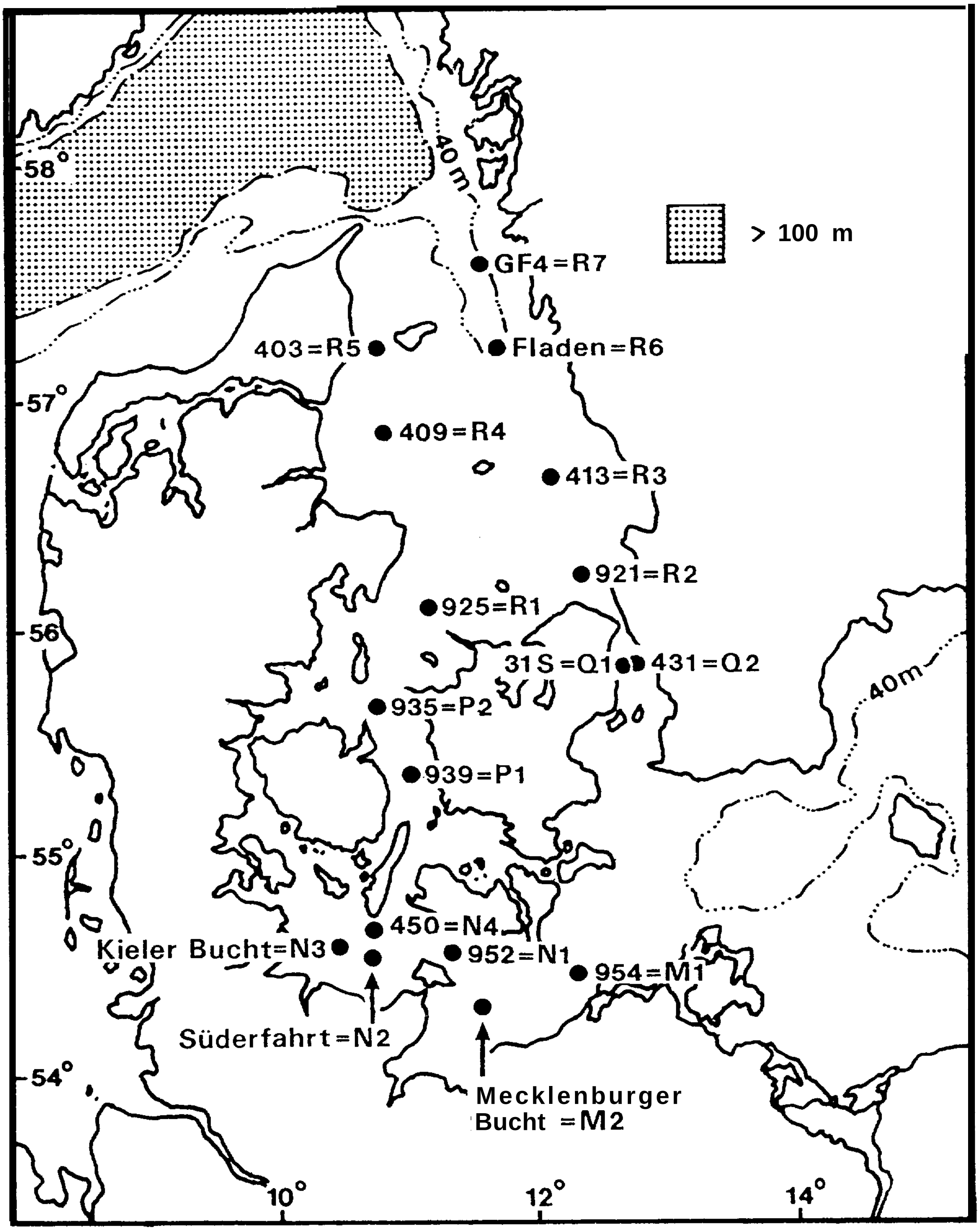

Figure 3. Monitoring stations in the Transition Area (Kattegat and Belt Sea)

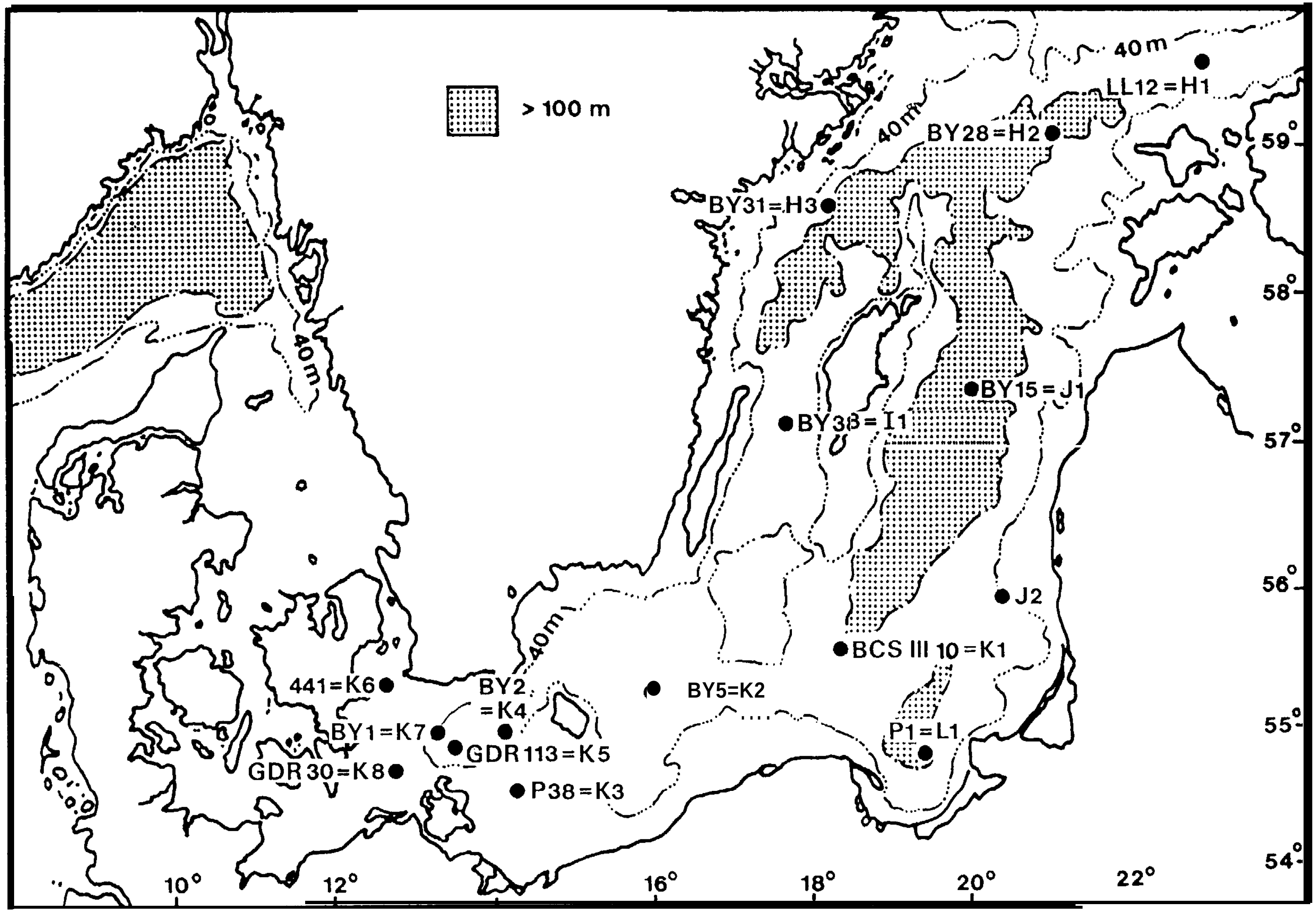

Figure 4. Monitoring stations in the Baltic Proper

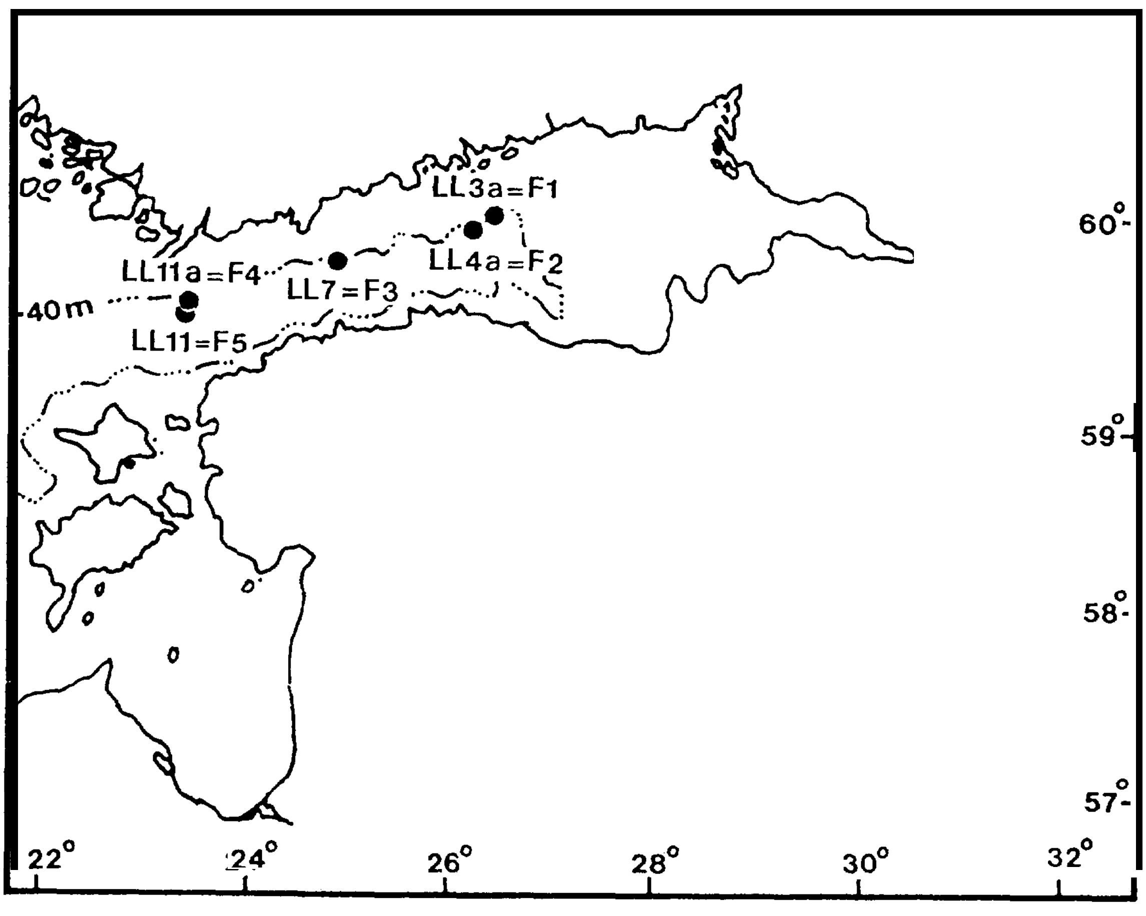

Figure 5. **Monitoring stations in the Gulf of Finland**

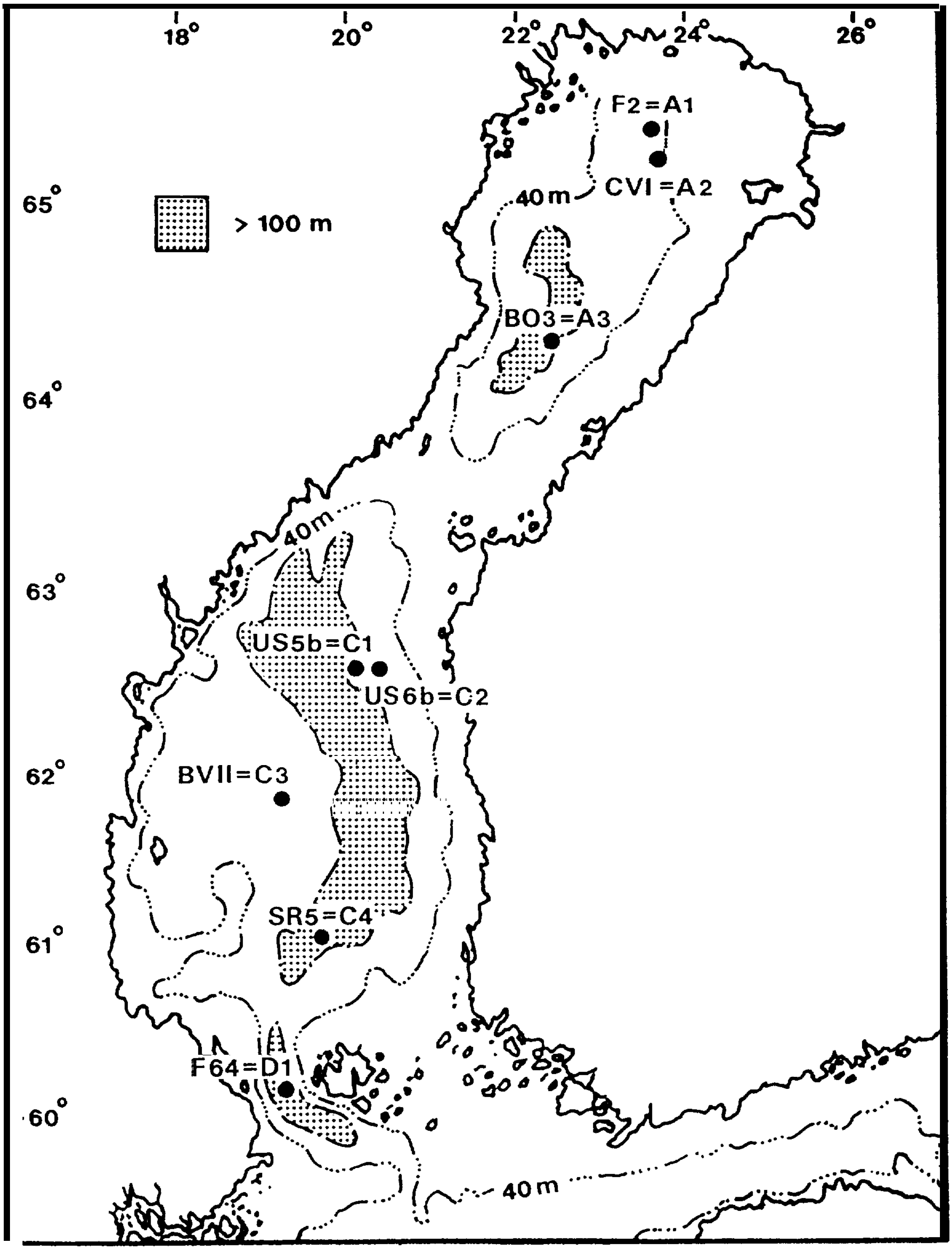

Figure 6. Monitoring stations in the Gulf of Bothnia

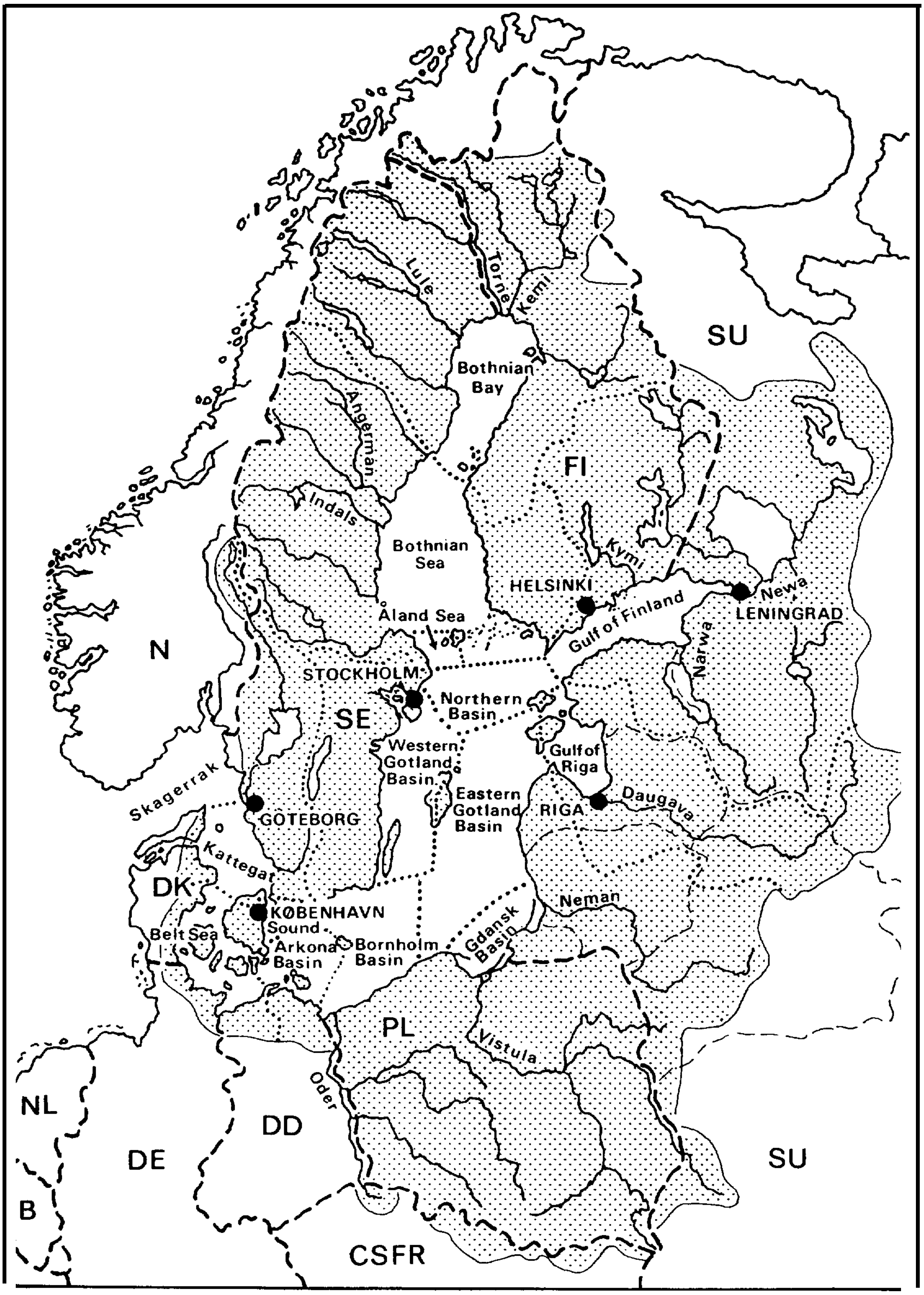

Figure 7. Sub-regions of the Baltic Sea as used in the Second Periodic Assessment

Baltic Sea Environment Proceedings 35B (1990)
Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1984-1988; Background Document

1. HYDROGRAPHY

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ABSTRACT

In this chapter the meteorological and hydrological conditions during the period 1984-1988 are described and their long-term variations are discussed.

The meteorological conditions during the assessment period can be characterized as variable. There have been three very cold and weak-wind winters (1984/1985 - 1986/1987) with heavy ice conditions followed by two warm and windy winters. At the beginning of the assessment period the summer seasons were rather cool, poor in radiation and weak in winds, but the summers of 1988 and 1989 were warm and calm.

The fresh water run-off to the Baltic Sea was higher than the long-term mean; only in 1985 and 1986 the run-off was close to that mean.

The most important phenomenon in the hydrography of the Baltic Sea during 1984-1988 was the continuing decrease in salinity in nearly all regions and layers. This process is mainly caused by the lack of major inflows of highly saline water during the last 13 years. The small inflows in spring 1986 and fall 1988 have had only effects on salinity and temperature in the deep layers of the Arkona, Bornholm and Gdaiisk Basins.

Temperature and density in the deeper layers of the Eastern Gotland Basin have also been decreased considerably and the halocline and isohaline depths have been noticeably descending. Depending on diminution of vertical density gradients the vertical mixing processes between the different layers have been enlarged.

The current stagnation period, at least in the Eastern Gotland Basin, must be regarded as one of the largest and most serious stagnation intervals ever recorded during this century and this has caused such extreme changes in the deep layers that never have been observed since the start of oceanographical observations in the Baltic Sea.

1.1 METEOROLOGICAL, ICE AND WATER EXCHANGE CONDITIONS

J. Launiainen[8] and T. Vihma[8]

1.1.1 Weather and ice conditions

As experienced, the 1980s have been a time of considerable variations in weather. This has been found in air and water temperatures as well as in sunny or cool summers, in wintertime ice and wind conditions, in wet and dry periods etc. In the following, some of these features are considered in more detail in the light of monthly anomalies from a couple of representative stations in the Baltic Sea area.

From monthly air temperature anomalies, given in Fig. la for Bornholm and Russarö (an island off Hanko), we can find a general decrease in air temperatures from 1982 to the end of 1987. Especially during 1985-1987 the mean air temperature anomaly was negative when compared with the normal 1931-1960 period. The cold winters of 198111982, 1984/1985 and especially the one of 1986/1987 are distinctly seen. On the contrary, the winters of 1982/1983, 1987/1988 and especially the 198811989 (and 1989/1990) were warm. Quite generally, the period from 1987 up to now (1990) has been rather exceptional. The air temperature variations above (Fig. la) were seen to be reflected in sea water temperatures as well, as revealed in various area1 sub-reports in this chapter.

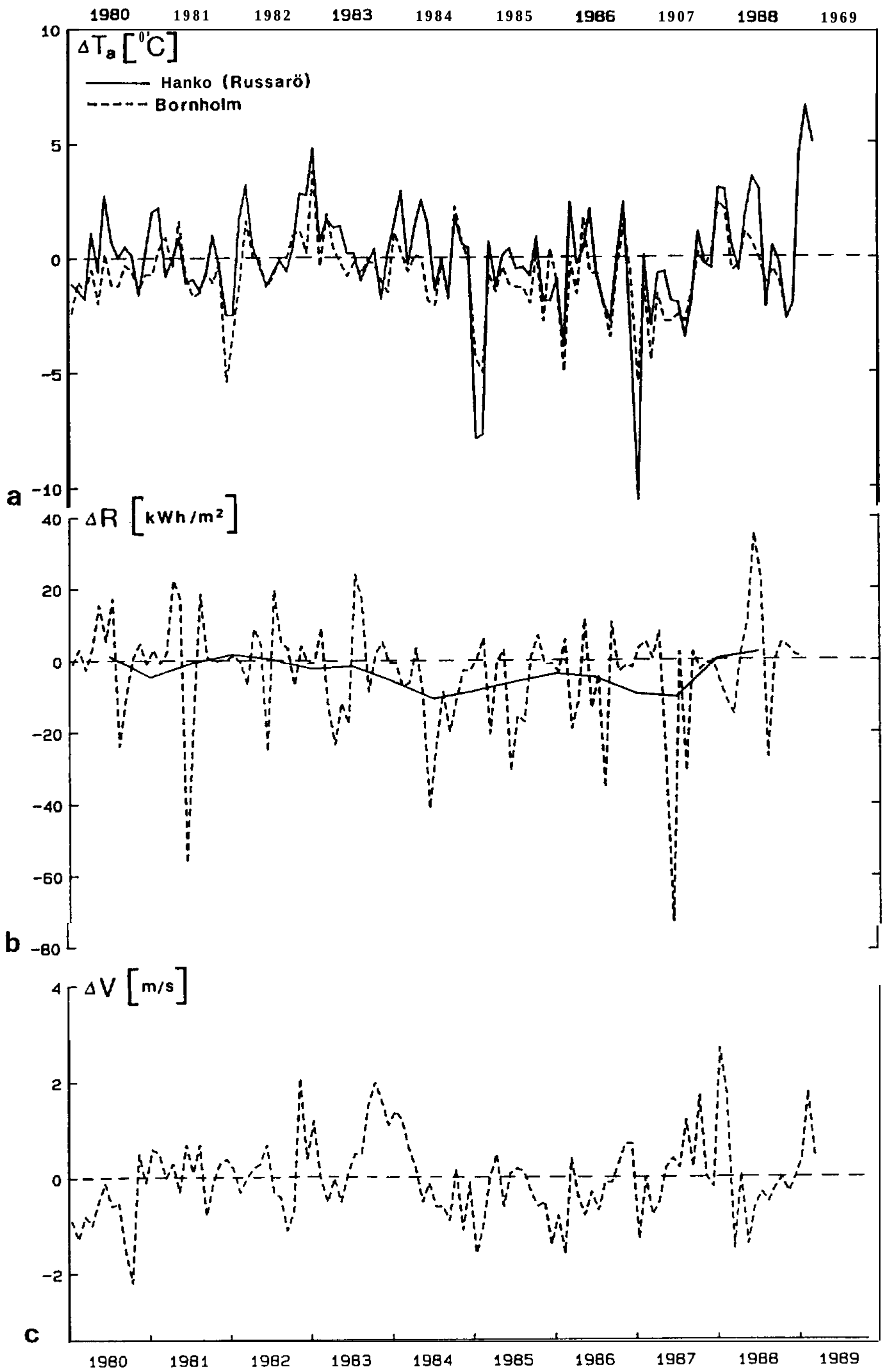


Figure 1 a. Monthly air temperature anomaly, compared to the 1931-1960 mean, at two stations (Bornholm and Hanko) in the Baltic Sea area during 1980-1989.

b. Monthly global radiation anomaly, compared to the 1961-1980 mean, at Bromma, Stockholm (dashed line). The solid line gives one year floating averages.

c. Monthly surface wind speed anomaly, compared to the 1961-1980 mean, during 1980-1989 at Russarö (Hanko).

(Data from the monthly reports by the Finnish Meteor. Inst., by the Swedish Meteor. and Hydrol. Inst., and by the Danish Meteor. Inst.)

From Figure lb, which gives the global radiation anomaly for Bromma (Stockholm), it is interesting to note that during this decade there was a decreasing trend in solar radiation, up to the very sunny summers of 1988 and 1989. Especially during the summers from 1984 to 1987 the radiation anomaly was negative.

The wind speed anomaly in Figure lc shows that after the positive anomalies of the windy period of 1982-1983, the middle of the 1980s was time of negative anomaly, i.e. time of light winds, up to the windy seasons of 1987 and 1988.

The ice conditions for the winters covering the last decade may be summarized as given in Table 1 below.

Table 1. Ice conditions for the winters 197811979 to **1989/1990.** Max coverage [km^2] gives the annual maximum area covered by ice, given as percentage of the total Baltic Sea area in the next column. "category" **characterizes** the severeness of the winters, mainly as referred to the long-period mean annual maximum ice coverage (of 193 000 km^2 for the winters 1890-1989).

winter	max coverage / km^2	% of Baltic Sea area	"category"
1978/79	325 000	78	quite severe
1979/80	260 000	63	quite normal
1980/81	175 000	42	normal
1981/82	255 000	61	normal
1982/83	117 000	28	quite mild
1983/84	187 000	45	normal
1984/85	355 000	86	severe
1985/86	337 000	81	severe
1986/87	405 000	98	extr. severe
1987/88	149 000	36	mild
1988/89	52 000	13	extr. mild excl. the northermost parts of the Gulf of Bothnia
1989/90	67 000	16	

As a summary, we may see that during the 1980s there has been three very cold and calm winters between 198411985 and 198611987 with heavy ice conditions and three warm and windy winters of **1987/1988**, 198811989 and **1989/1990** (cf. Fig. 5 also). As to the summer weather, the summers in the middle of the decade were rather cool, radiation poor and also calm. Additionally, during the last three years (1987-1989) the variations in temperature, global radiation as well as in wind have been very prominent. In this respect, one should note, e.g. that the very windy (sedimentous materia remobilising) autumn and winter of 198711988 were followed by a calm, radiation-rich and warm summer of 1988; a set of physical events favouring extraordinary blooms and unpleasant biological and environmental events happened in the sea during 1988.

1.1.2 Fresh water run-off and water exchange

Table 2 gives the estimates of the yearly total fresh water run-off into the Baltic Sea since 1971, and the graph of the long-term time series of the run-off is given in Figure 2. The recent estimates since 1971 are regression estimates using weighted yearly means of two Finnish rivers discharging into the Baltic Sea. The method is described in more detail in the First Periodic Assessment (Launiainen et al. 1987).

As suggested by the data, the fresh water run-off (surplus) seems to have been rather large during all the 1980s. Only during the cool years of 1985 and 1986 the run-off was close to the long-period average but after 1980 none of the years has been a "dry" one. By mutual comparison of the various decades in Fig. 2 the "wetness" of the 1980s is apparent, as well.

For estimating the sea water exchange and the most prominent occasions of inflow and outflow, between the Baltic Sea and the North Sea, volume changes (sea level) of the Baltic Sea were studied, following the approach given in the First Periodic Assessment (Launiainen et.al., 1987). Accordingly, the volume change estimates were calculated by a simple method suggested by Jacobsen (1980), using the filtered daily sea level data from Degerby, Finland (obtained from the Finnish Inst. of Mar. Res., Helsinki). The method seems to yield rather accurate first order estimates for the volume variations, on the basis of which the sea water exchange may be estimated, using the equation of

$$dV/dt = Q_o + Q$$

where dV/dt is a change in the volume (sea level), and Q_o is the fresh water surplus, and Q is the net sea-water exchange.

Figure 3 gives the Baltic Sea volume variations during 1980-1988 and suggests the situations of the most prominent potential inflows and outflows. Fig. 3 also lists the rates of the accumulated yearly sea water exchange (Q). Those have been calculated according to the equation above, for the yearly fresh water surplus using an approximation of Q_o ~ Q,,, where Q_{BT} is the total fresh water run-off from Table 2. This latter approximation includes the common assumption that precipitation and evaporation above the Baltic Sea balance each other.

The results in Figure 3 show, e.g. that the volume variations of the Baltic Sea were somewhat more pronounced during the first years of the 1980s than during the last few years since 1985. A similar tendency may also be found in the sea water exchange. Accordingly, the time history since 1985 suggests less prominent inflows and outflows, compared with the late 1970s and the first years of this decade. This seems to be the case, e.g. even for the autumn and winter 198711988, no matter how windy it was. On the other hand, most of the inflows in the beginning and middle of the 1980s, identified by the increased deep water salinities in the Bornholm Basin (section 1.232 in this chapter), coincide with the potential inflow situations suggested in Figure 3.

Table 2. Estimated total fresh water run-off Q_{BT} (in m^3/s) into the Baltic Sea. Annual means during 1971-1988. M_{21-70} and SD_{21-70} are the observed mean and standard deviation of 1921-1970 (Mikulski, 1980). The estimates given by the formula $Q_{BT} = 5.36 \cdot (1.6 \cdot QV + QK) + 7145$, where QV and QK are the yearly mean discharges of Vuoksi and Kemijoki, respectively. Q_V and QK data from the Hydrol. Office, Nat. Board of Waters and Env., Finland.

	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980
Q_{BT}	14230	13900	14370	15520	16960	13550	14750	13670	13690	14030
	1981	1982	1983	1984	1985	1986	1987	1988	M_{21-70}	SD_{21-70}
Q_{BT}	17560	17270	16280	16400	15160	14920	16440	16840	15005	1790

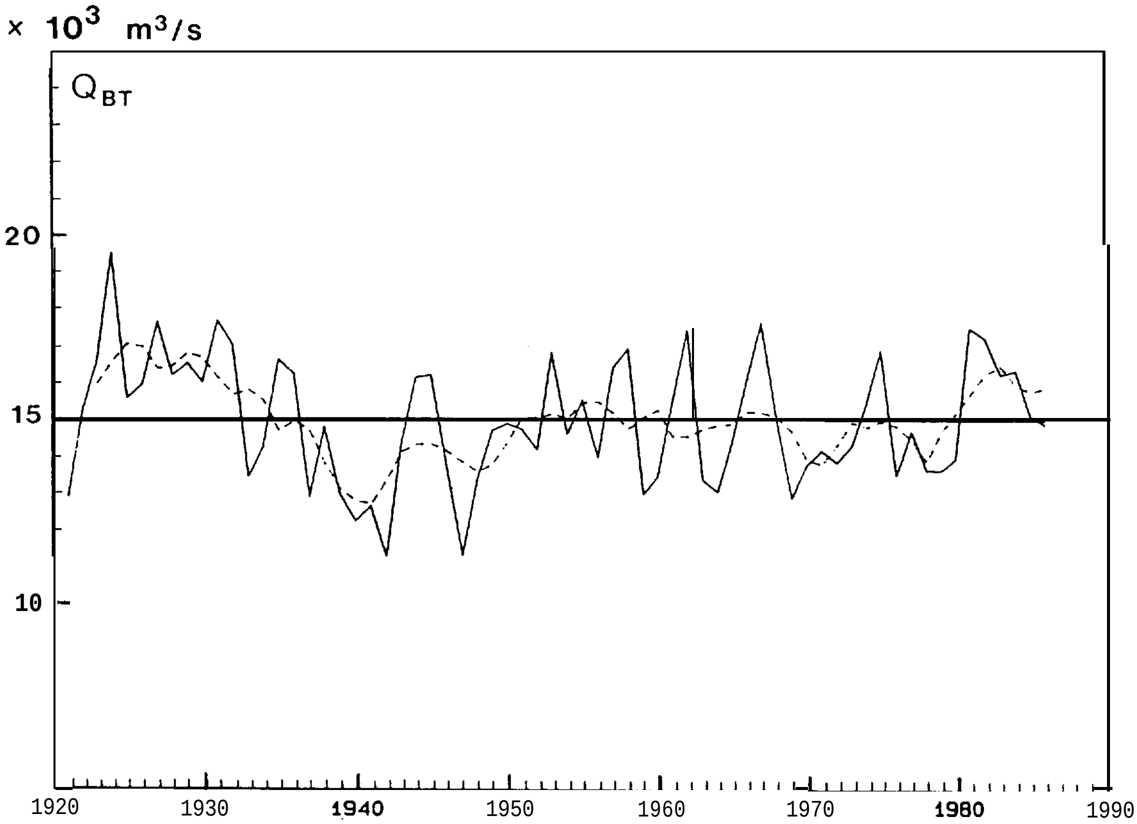


Figure 2. Total fresh water run-off into the Baltic Sea. Annual means. Data for 1921 to 1970 (observed) from Mikulski (1980), and onwards 1971 as estimates from Table 2. The dashed line represents 5 year floating averages.

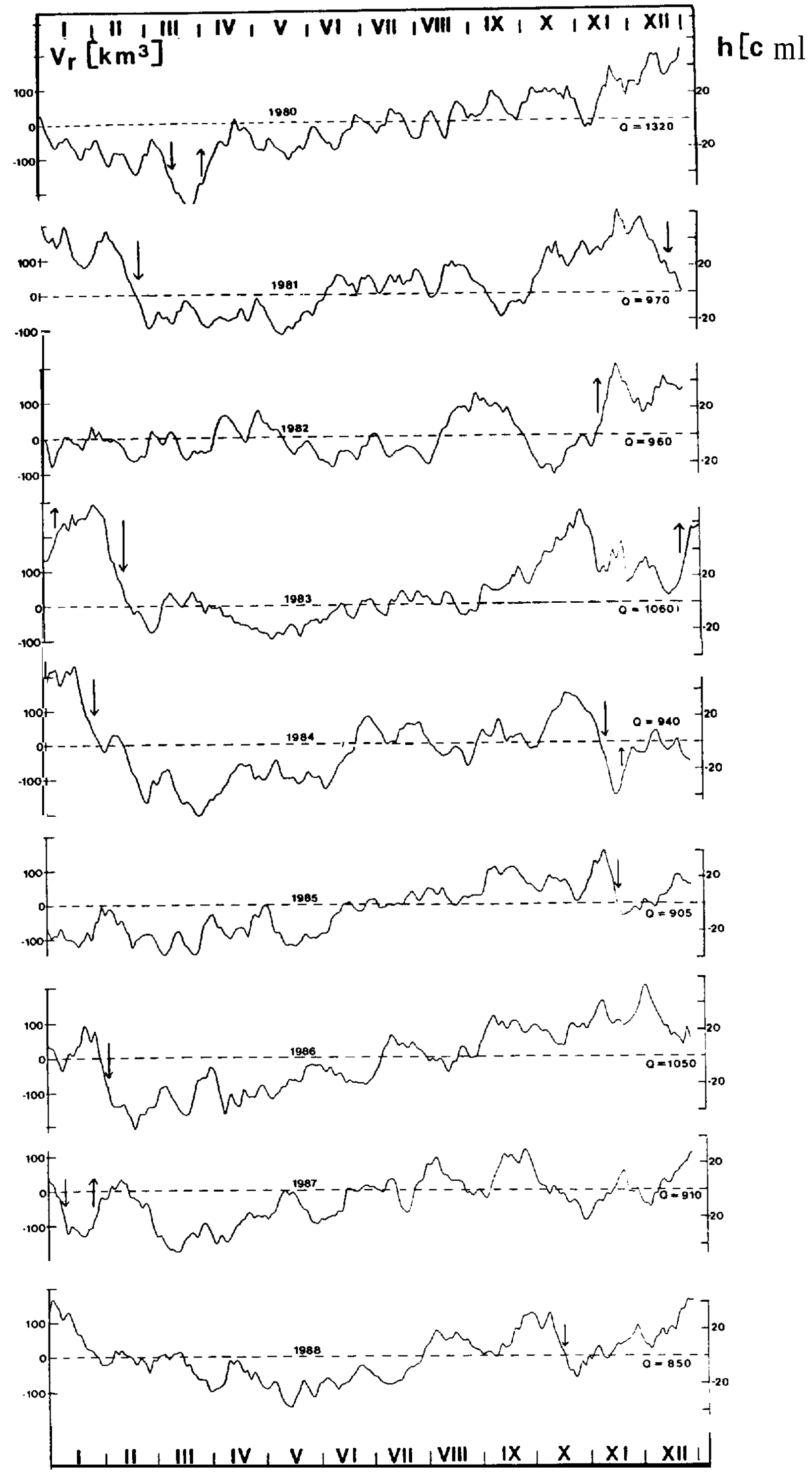


Figure 3. Water level (h) and volume ($V_r = V - V_{aver}$) variations of the Baltic Sea during 1980-1988, based on daily mean sea level data. The arrows indicate the most probable inflow and outflow situations, between the Baltic Sea and the North Sea. Q gives the yearly accumulated sea water exchange (yearly net inflow ~ yearly net outflow).

As a conclusion, we may see that the fresh water surplus (run-off) has been rather large during all the 1980s. In the surface layers of the Baltic Sea this means in a way a larger surface water exchange, as well. Accompanied with a decrease in the sea water exchange, on the other hand, the fresh water surplus is supposed to have been the cause of a decreasing trend of the salt balance of the whole Baltic Sea. Additionally, only **minor inflows of salt-rich North Sea water have occasioned during the latest years (Fig. 3).** From the important point of view of a renewal of bottom waters in the deep basins of the Baltic Sea, this is supposed to be the main reason that no such a renewal has happened and **the** stagnation period of the deepest bottom waters has lasted over 13 years, so far. This is the case although proper physical conditions for a renewal have prevailed in the deep basins for over several years (cf. Baltic Marine Environment Protection Commission, 1987), owing to the very low density of bottom waters.

1.1.3 Long-term changes

Various long-term features of the Baltic Sea hydrography and climatology were discussed in the First Periodic Assessment. Climatological air-sea temperature coupling, long-term variations in sea level changes, long-term decline in annual ice covered season, and correlation between the fresh water run-off and Baltic Sea salinity were considered. For this context, three figures were adopted, the monitoring period data added. Figures 4 and 6 indicate the close climatological air-sea temperature coupling and the long-term variation in annual ice covered season, and the rough test examples in Figure 7 suggest an interesting correlation between the estimated fresh water run-off and the Baltic Sea salinity. For further interpretation and discussion, see the First Periodic Assessment (Launiainen et al. 1987). Additionally, the long-term variations in the yearly maximum ice coverage of the Baltic Sea are shown in Fig. 5.

Presently, the expected change in global climate has gained increasing daily attention. A prognosis for a global greenhouse warming up has relevant physical arguments, and various estimates, e.g. for mean latitudinal temperature increase, have been done. In global scale, the climate especially during the 1980s has been seen as a sign of this warming up. In this meaning, it is interesting to consider the case, if not otherwise but on the local basis.

The air temperature during the 1980s does not indicate any increase up to 1988 (Fig. la); merely the case is an opposite one, during which the global radiation (Fig. lb) shows a decreasing trend, as well. This kind of a recent decreasing trend in the air temperature was also found in an extensive data set over Sweden (Alexandersson and Eriksson, 1989). However, when comparing e.g. the 1972-1988 air temperature anomaly with the "normal" 1931-1960 period, both the Bornholm and Russarö data showed the latter period, especially the summer seasons to have been cooler than the normal period. Because of a close air-sea coupling (cf. Fig. 4) the above is valid, most probably, for the water temperature also. On the other hand, in the light of the annualmaximum ice cover, the late 1970s and 1980s (up to 1988) have been time of moderate or severe ice winters (Fig. 5 and Table 1).

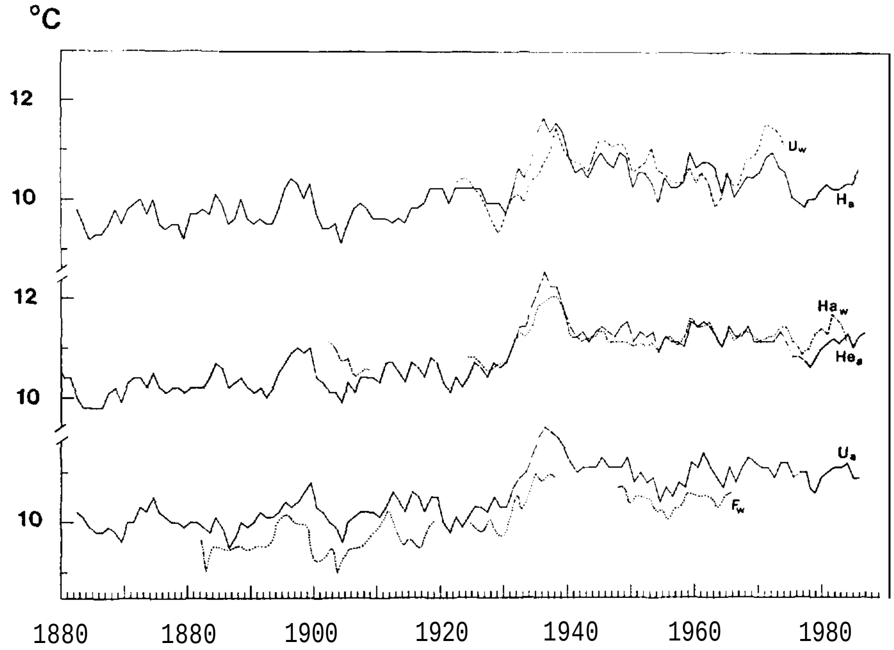


Figure 4. Long-term variation of air temperature and sea surface temperature (5-year running averages). Redrawn from Launiainen et al. (1987) adding the monitoring period data.

U_w = water temperature at Ulkokalla (June-Ott)
H_a = air temperature in Haparanda (June-Ott)
Ha_w = water temperature at Harmaja (June-Nov)
He_a = air temperature in Helsinki (June-Nov)
F_w = water temperature at Finngrundet (June-Nov)
U_a = air temperature at Uppsala (June-Nov)

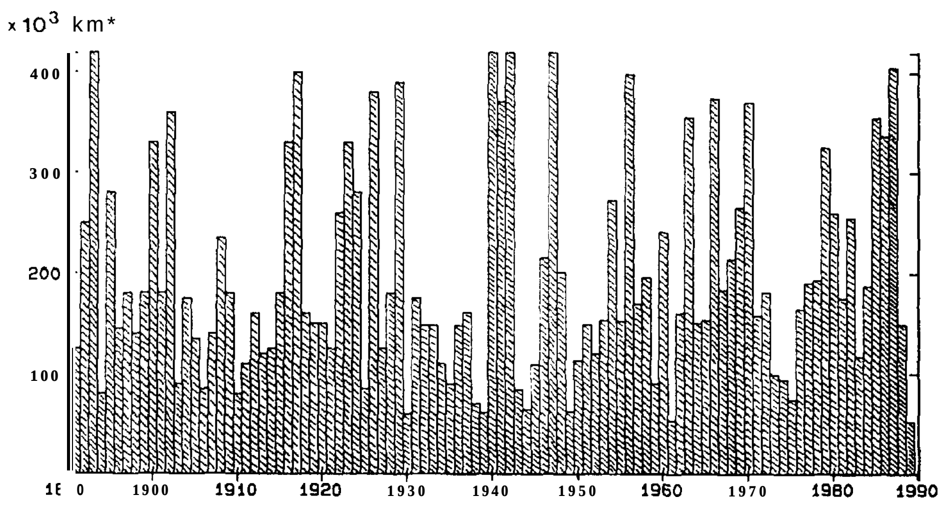


Figure 5. Time series of the yearly maximum ice extent in the Baltic Sea during 1890/1891 - 1988/1989. Area of 420 000 km^2 corresponds to the total area of the Baltic Sea. (From the Finnish Inst. of Mar. Res..)

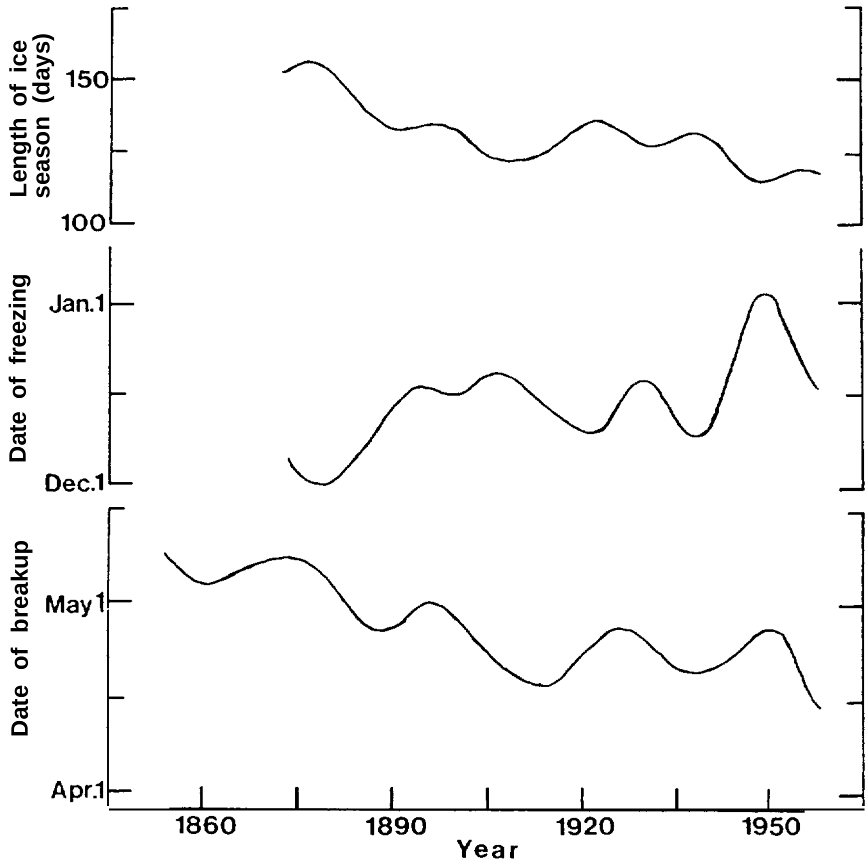
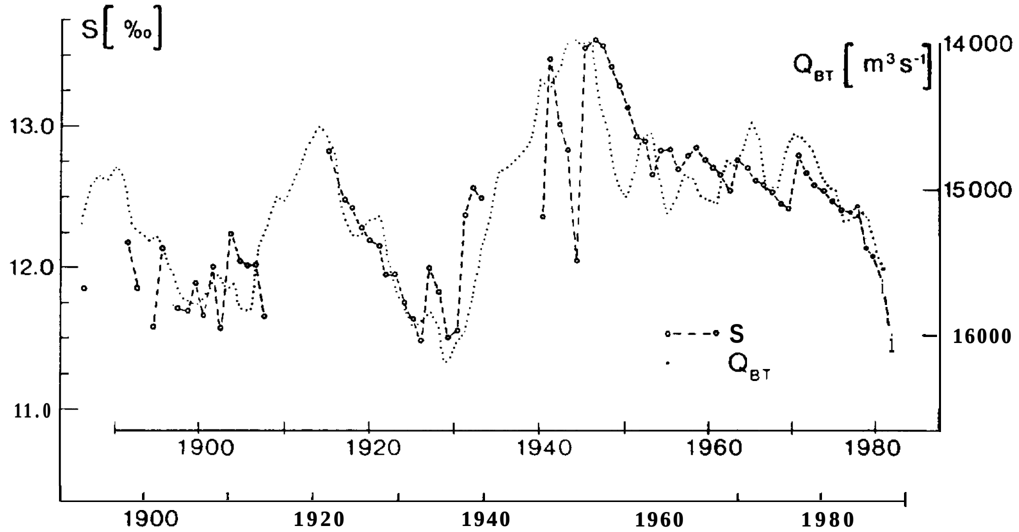


Figure 6. Time series of low pass filtered data of freezing, ice breakup, and the length of ice season at Helsinki. Redrawn from Leppäranta and Seinä (1985).

a


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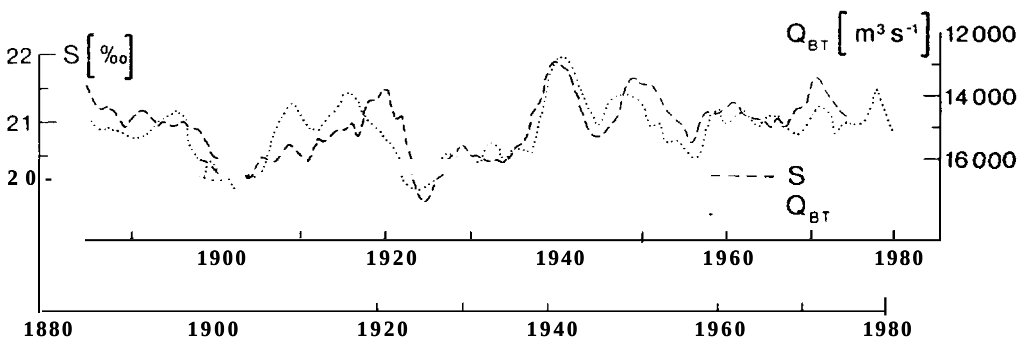


Figure 7 a. Long-term variations of the deep water (200m) salinity in the Gotland Deep (BY 15 = BMP 11) in comparison with the estimated fresh water run-off to the Baltic Sea. For the best fit, the time axis of the low pass filtered (15-year running average) run-off have been shifted forward 6 years. Redrawn from Launiainen et al. 1987 adding the monitoring period HELCOM data.

b. Long-term variations of the surface salinity at Anholt (5-year running averages from Malmberg and Svansson, 1982) in comparison with the estimated total run-off to the Baltic Sea (S-year running averages from Launiainen, 1985).

The river run-off into the Baltic Sea turned out to have been rather large during the 1980s. Physically, this is coherent e.g. with the data given by Alexandersson and Eriksson (1989) according to which the precipitation in Sweden during the 1980s seems to have been larger than during a couple of the previous decades.

Summarizing, from a local point of view, it seems that one cannot detect any reliable sign of excess warming up from the time series of mean temperatures of the Baltic Sea area, not at least until 1988. *In* 1988-1990, on the contrary, very warm summers and mild winters prevailed. One can hardly see, however, whether the recent prominent variations in temperatures and other meteorological and hydrographic quantities reveal signs of a global climatic change or not.

1.1.4 **Long-term changes in transparency**

The changes in transparency according to Secchi depths as an integral parameter for the conditions in the surface layer in the Baltic Sea were studied in some papers during the last two years.

Launiainen et al. (1989) compared the transparency observations in the Northern Baltic Sea made during 1914-1939 with those of the last two decades of 1969-1986, taking into account the methodical differences. The following conclusions were made (cf. **Fig.** 8):

> The present Secchi depth (transparency) shows pronounced decrease (2.5 to **3m,** generally) compared with that during the first half of the century.
> The distribution of the Secchi depths **has become** *more* **narrow, so** that large Secchi depths **(> 14m) are very seldom, if any.**

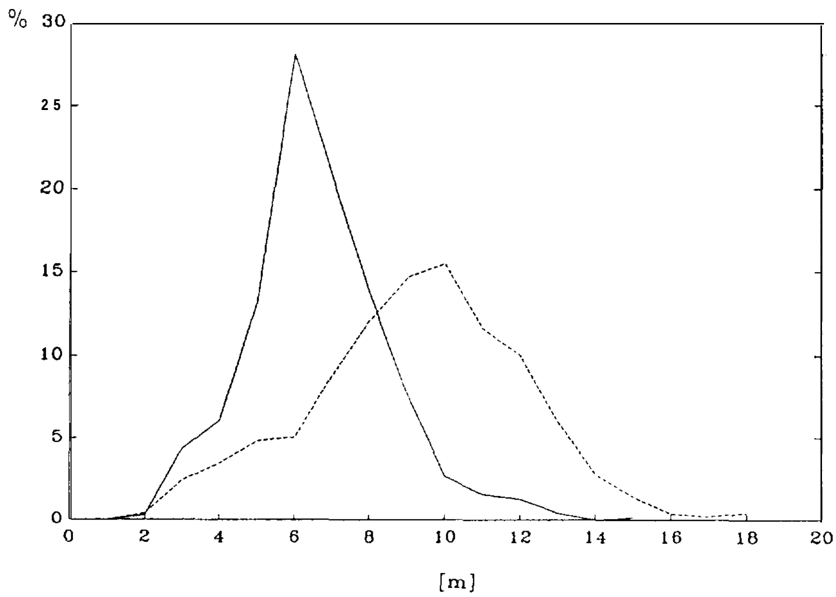


Figure 8. Frequency distribution of Secchi depth (in m) in the Northern Baltic Sea (in %) for the period of 1914-1939 (broken) and for the period of 1969-1986 (continuous line). From Launiainen et al. (1989).

Analogously, in a reference area on the Swedish coast of the Baltic Proper, not influenced by point pollution sources, the Secchi depth was found to have been decreased by 4m since 1965 (Persson, 1990; Cederwall and Elmgren, 1990).

1.2 HYDROGRAPHICAL CONDITIONS

1.2.1 The Kattegat

S. Carlberg[4]

The hydrographical development in the Kattegat is demonstrated with data from the representative station Anholt E (413 = BMP R3).

The overall picture for the Kattegat is the traditional one with high variability in temperature and salinity.

The temperature shows a clear and strong variation in an annual cycle not only at the surface but also at 30 m and in the bottom layer (see Figures 9-10). The unusually cold winters of 198411985 and 1986/1987, see Figure 6, are certainly reflected as minima at 30 m and below, but the observed minima are not significantly different from other years. The winter of 1981/1982, e.g., was cold but not that cold as 1984/1985 but seemed to have produced a lower minimum. It is interesting to note that when the winter is mild, e.g. 1982/1983 and 1983/1984, the temperature minima in the intermediate and bottom waters seem to appear earlier in the winter as compared to the effects of the cold winters.

Concerning salinity, however, the variability largely follows an annual cycle, both at the surface and in the deeper layers (Figures 11-13). The inflows of higher saline water that occurred to the Baltic Proper at the turn of 1982-1983, in the spring of 1986 and in September 1988, can be correlated to annual increases of salinity in the Kattegat surface and bottom waters as shown in Figures 11-13. It is important to note, however, that these increases do occur every autumn or winter but they do not automatically cause an inflow of water to the Baltic. When an inflow does occur the Kattegat salinity may be rather high in the surface as in 1982/1983 (Figure 11) but not necessarily as in 1985/1986 (same Figure) and in the deeper waters these events are not significantly different from other years (Figures 12-13). The various effects of these inflows are described in the following sub-chapters.

The changes of temperature and salinity are summarized in Table 3.

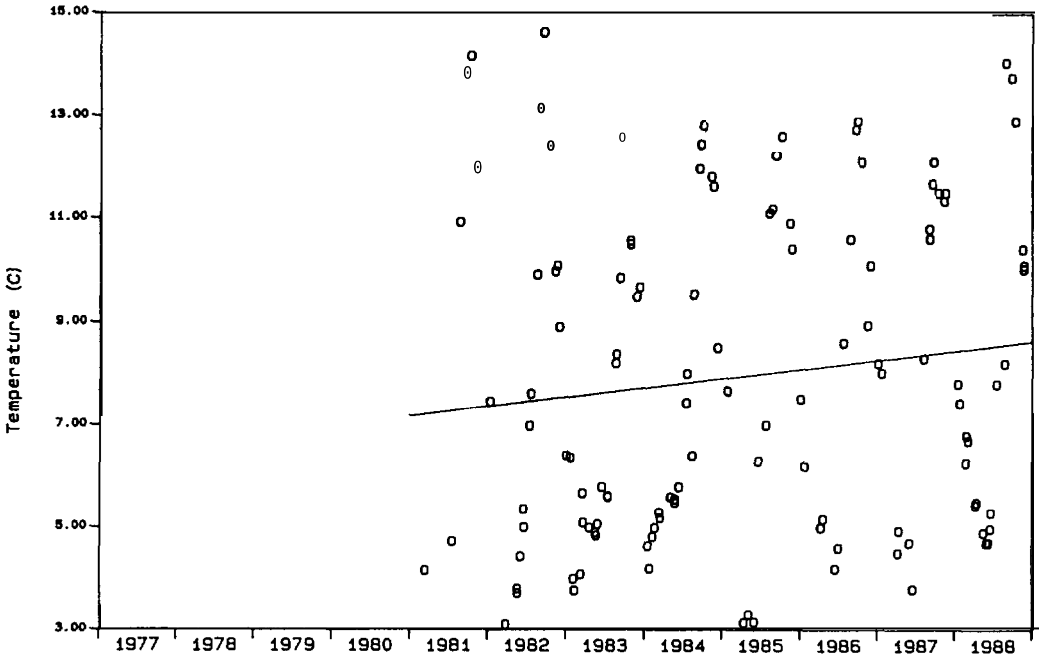


Figure 9. Temperature variation at the station Anholt E (413 = BMP R3), in 28-32 m water depth.

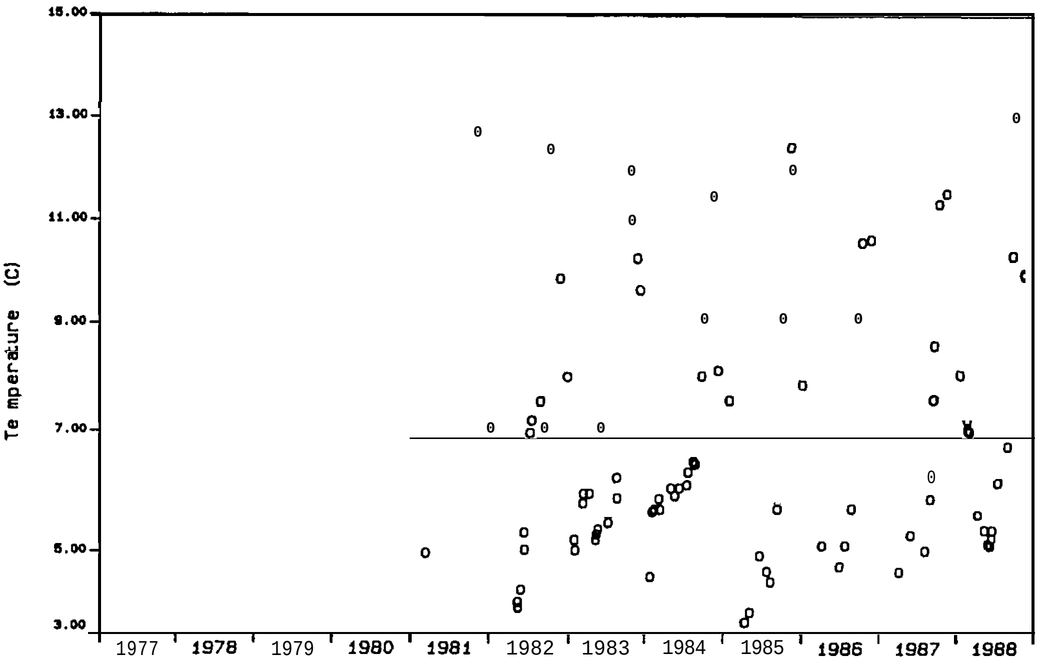


Figure 10. Temperature variation near the bottom (48 m to bottom) at the station Anholt E (413 = BMP R3).

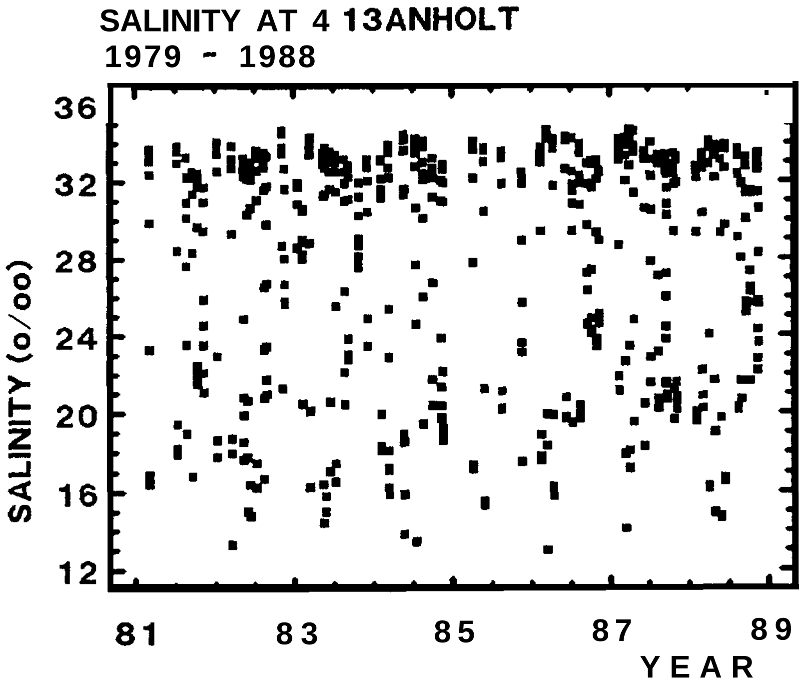


Figure 11. Salinity variation at the surface of the station Anholt E (413 = BMP R3).

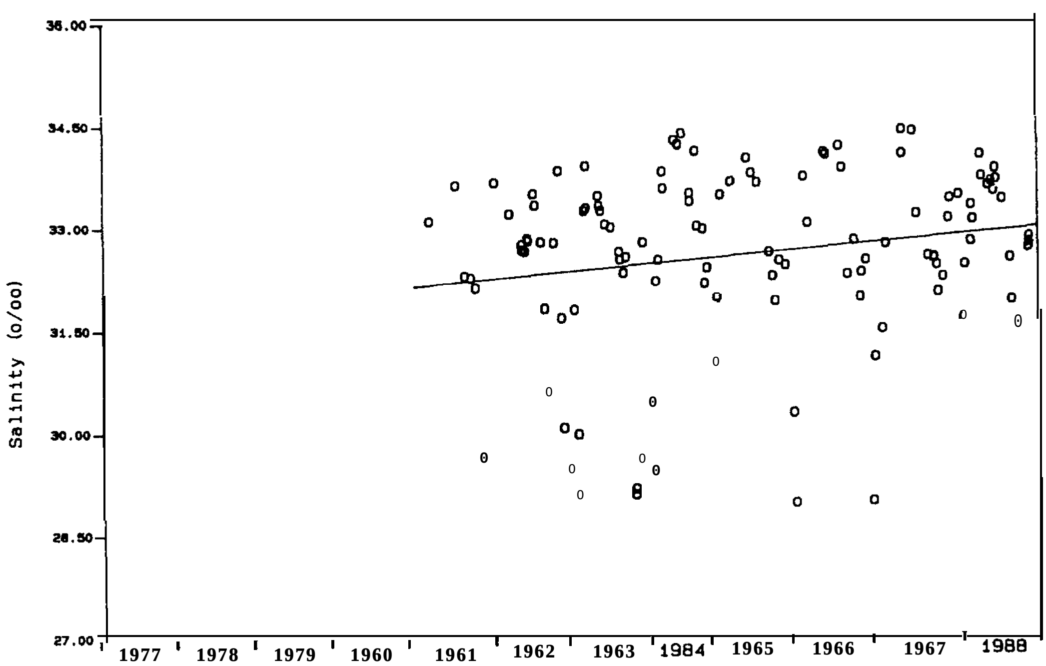


Figure 12. Salinity variation at the station Anholt E (413 = BMP R3) in 28-32 m water depth.

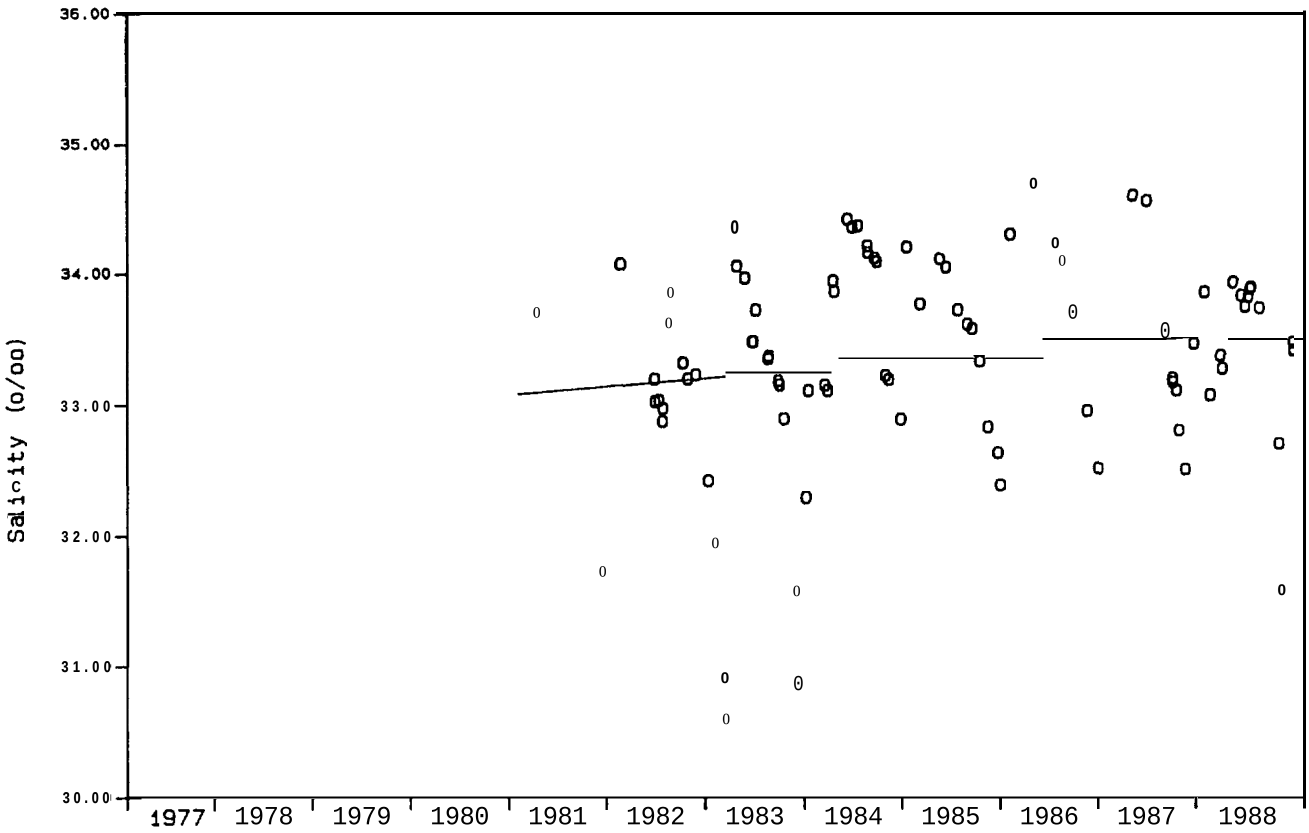


Figure 13. Salinity variation near the bottom (48 m to bottom) at the station Anholt E (413 = BMP R3).

Table 3. Characteristic changes of temperature and salinity in the Kattegat (Station Anholt **E** = 413 = **BMP R3).**

Parameter	Period	Depth	Overall trend	Annual changes
S	1981-1988	28-32m	+0.84 PSU	+0.11 PSU/a *
S	1981-1988	48m-bottom	+0.50 PSU	+0.06 PSU/a
T	**1981-1988**	**28-32m**	+1.48 °C	+0.19 °C/a
T	1981-1988	48m-bottom	+1.00 °C	+0.13 °C/a

*** PSU = Practical Salinity Unit =** ^{o}/oo

1.2.2 The Belt Sea
W. Lange[7]

To recognize any possible trend between 1950 and 1983 the annual means of the temperature and salinity measured at l/v Fehmarnbelt near surface, at 5 m depth, 20 m depth and near bottom are shown in Figures 14 and 15. But these time series do not come up to the time of the assessment period, because of great gaps in data after 1983. Therefore, the results may be only of general interest. In Figure 16 the annual vector means of the inflow and outflow observations at the surface of l/v Fehmarnbelt are shown covering the period 1950-1983.

More detailed information on the salinity stratification at l/v Fehmarnbelt between 1975 and 1983 is given by Weigelt (1987).

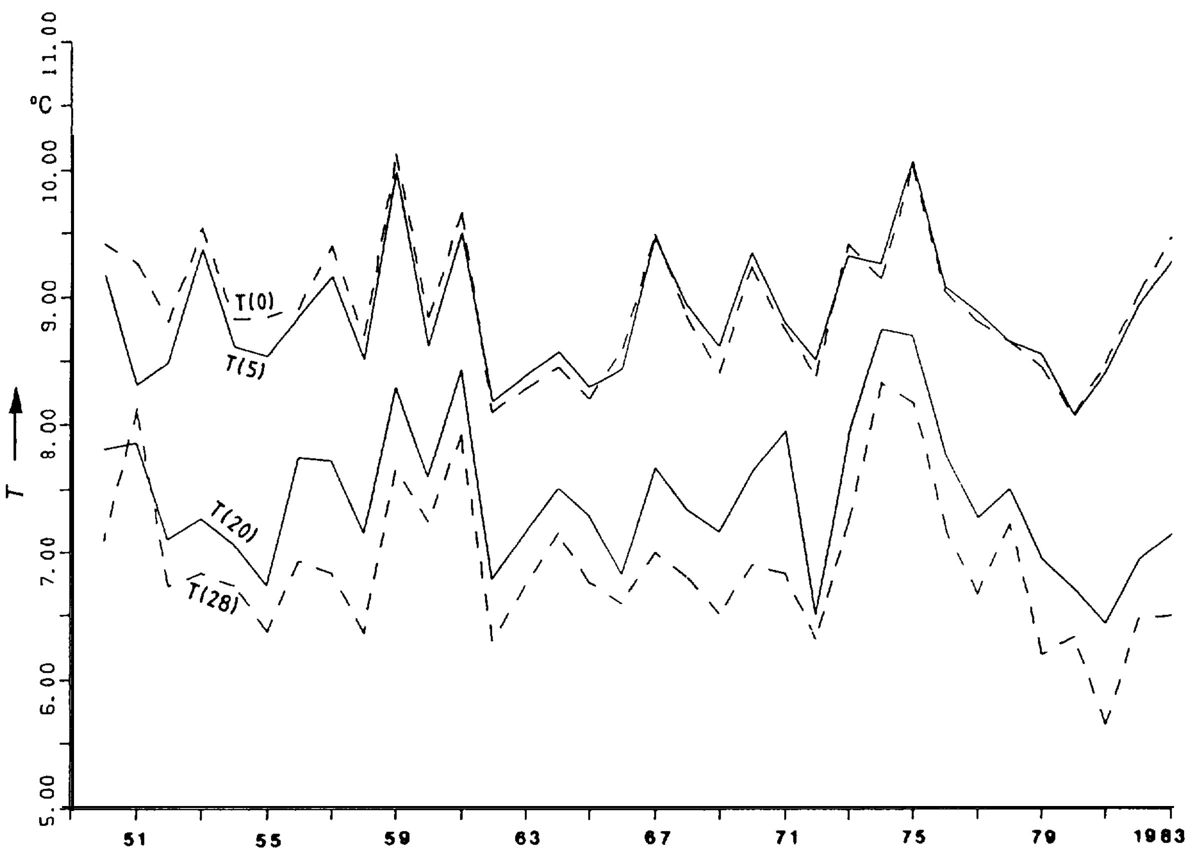


Figure 14. Annual means of the water temperature (°C) measured daily from 1950 until 1983 at lv Fehmarnbelt (BMP N1) near surface, at 5 m depth, 20 m depth, and near bottom.

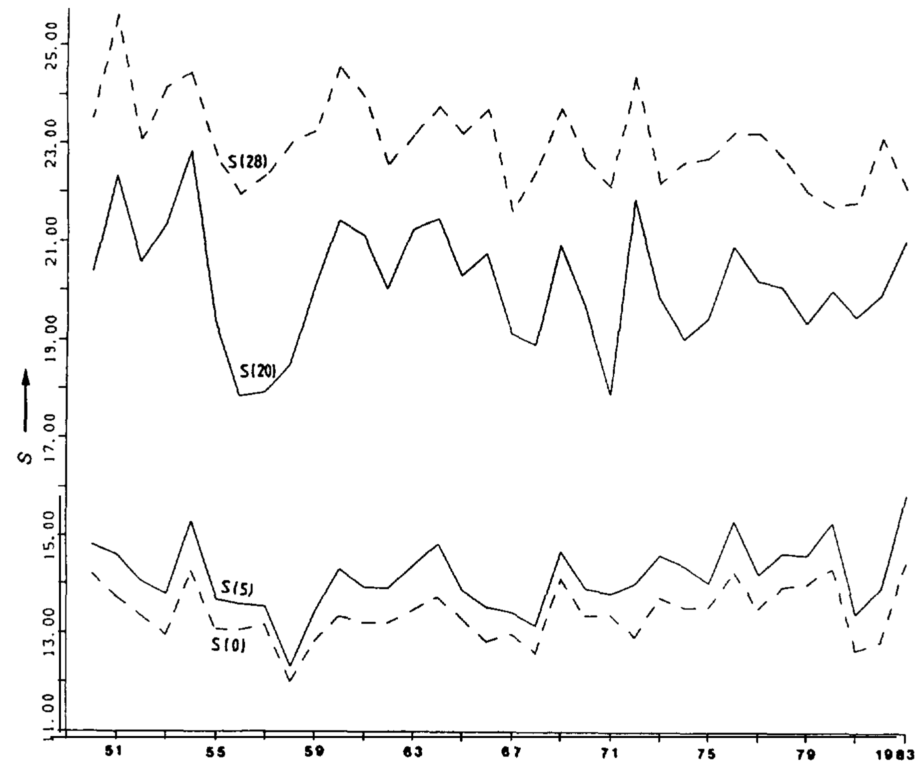


Figure 15. Annual means of the salinity (°/oo) measured daily from 1950 until 1983 at lv Fehmarnbelt (BMP N1) near surface, at 5 m depth, 20 m depth, and near bottom.

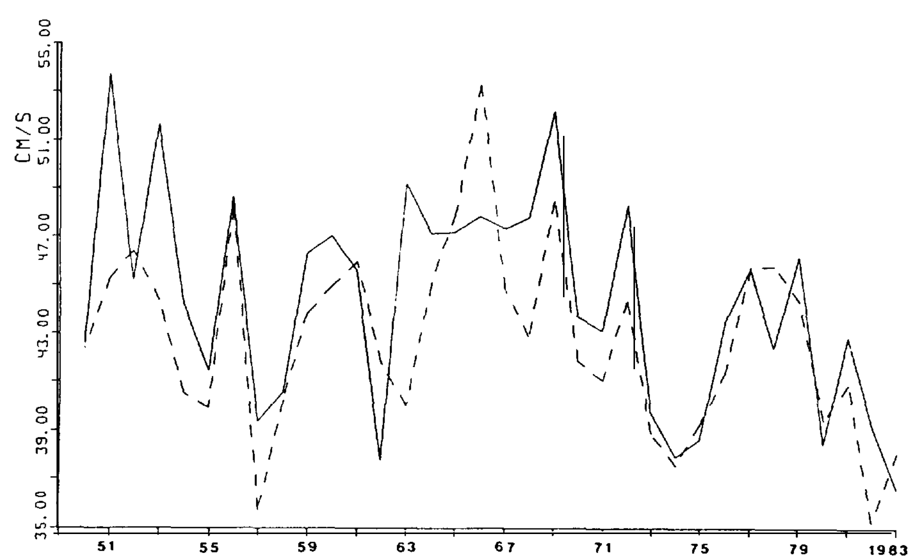


Figure 16. Annual vector means of surface current speed (cm/s) into (continuous line) and out of (broken line) the Baltic Sea measured at lv Fehmarnbelt (BMP N1) near surface from 1950 until 1983 once every 4 hours.

1.2.3 The Baltic Proper

During the period under review, inflows of higher saline water occurred at the turn of 1982-1983, in the spring of 1986 and in September 1988. The various effects of these inflows are described below.

1.2.3.1 Arkona Basin

W. Matthäus[2] and S. **Carlberg**[4]

The variations in the Arkona Basin are demonstrated with data from the representative station BY 2 (BMP **K4).** The general picture shows a high variability in salinity as well as temperature.

The temperature of the surface layer shows natural seasonal variations which are also present in the near bottom water where the effect of **advective** processes is superimposed on these variations. The variability is very high and there is no general trend in the temperatures. The winters of 198411985 to **1986/1987** were unusually cold in the entire area and this is clearly reflected even in the bottom layer where the temperatures reached almost **O°C** during certain periods.

The inflow that occurred at the turn of 198211983 increased the surface salinity by about 2 PSU, see Figure 17. After that the surface salinity decreases as a general trend. The mean decrease is about 1 PSU for the period and by the end of 1988 the mean is roughly the same as before the inflow. Superimposed on the general trend is the seasonal variation with winter periods with increased salinity. See also Table 4.

The near bottom water is dominated by a very high variability, see Figure 18. Despite the fact that more observations were available for the assessment period than for previous years, there is no evident trend in the salinity.

Table 4. Characteristic changes of temperature T and salinity **S** in the Arkona (BY 2 = BMP K4) and Bornholm Basins (BY 5 = BMP K2).

Area	Para-meter	Period	Depth	Overall trend	Annual changes
Arkona Basin	S	Nov 1982-1988	0- 2m	-0.8 PSU	-0.13 PSU/a
Bornholm Basin	S	Jan 1983-1988	0- 2m	-0.8 PSU	-0.13 PSU/a
Bornholm Basin	S	Jan 1983-1988	78-82m	-2.5 PSU	-0.42 PSU/a
Bornholm Basin	T	May 1980-Dec 1983	78-82m	+3.3 °C	+0.93 °C/a
Bornholm Basin	T	Dec 1983-Jul 1986	78-82m	-4.2 °C	-1.58 °C/a
Bornholm Basin	T	May 1986-1988	78-82m	+2.0 °C	+0.79 °C/a

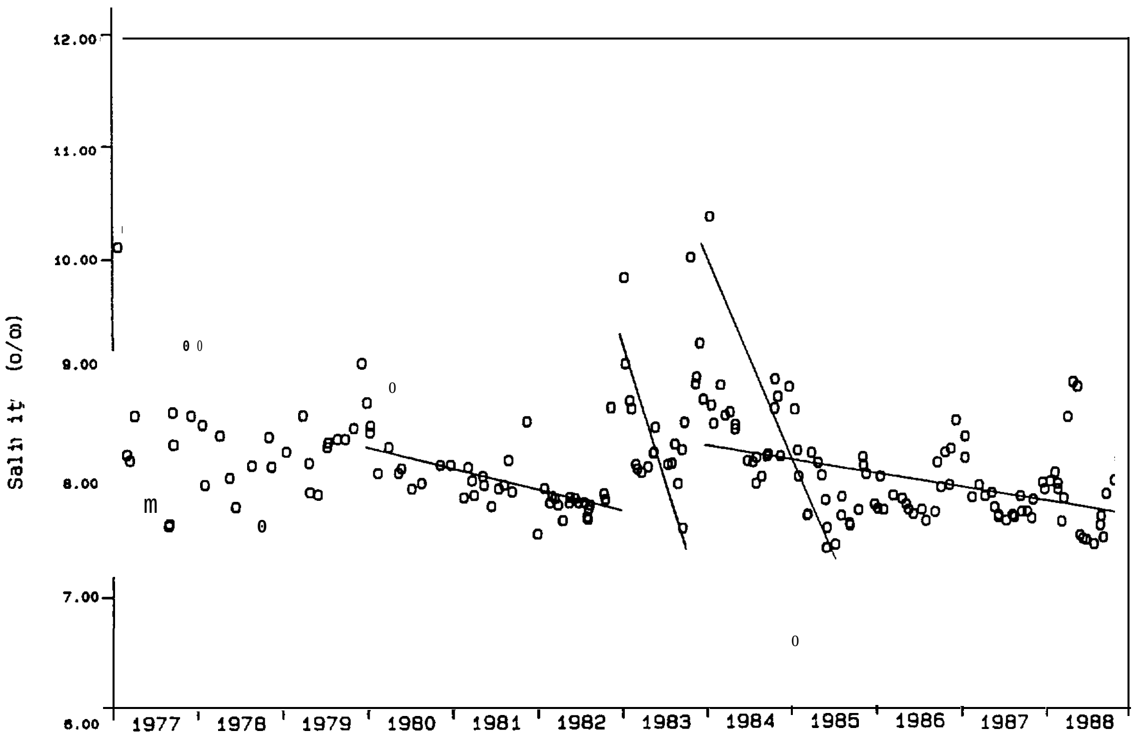


Figure 17. Salinity variation at the sea surface (0-2 m) in the Arkcna Deep (BY 2 = BMP K4).

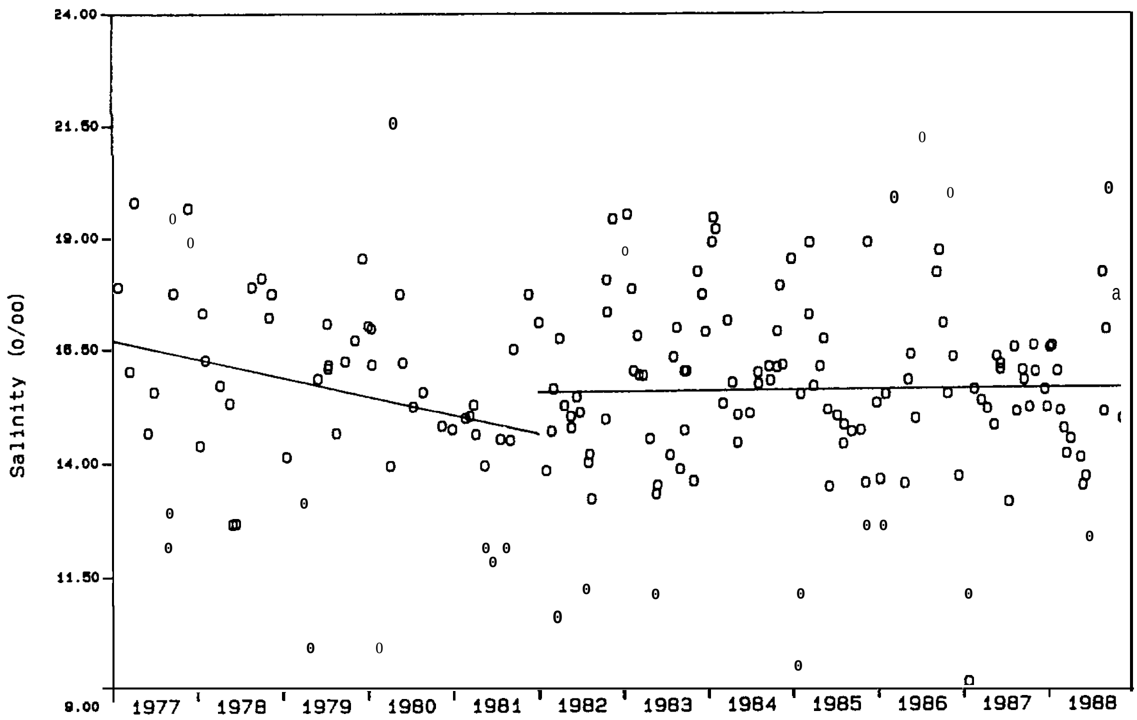


Figure 18. Salinity variation near the bottom (43 m to bottom) in the Arkona Deep (BY 2 = BMP K4).

1.2.3.2 Bornholm Basin

W. Matthäus[2] and S. Carlberg[4]

The variations in the Bornholm Basin are demonstrated with data from the representative station BY 5 (BMP K2).

The effect of the unusually cold winters referred to above are clearly seen also in the surface water of the Bornholm Basin. As an example, in March 1985, 1986 and 1987 the temperatures were below 0°C from the surface down to 30-50 m (Nehring and Francke 1987 a, b). This negative anomaly was conserved in the intermediate water until the autumn. The very cold winter and the rather cold and rainy summer of 1987 kept the temperature of the surface layer below what is normal (Carlberg et al. 1988).

Concerning the deep water a clear change took place. After a period of mainly increasing temperature by about 6 °C in 1980-1983, the temperature decreased rapidly by about 5 °C in 1984-1986, see Figure 19 and Table 4. After that the effect of the inflow of relatively warm water in 1986 and 1988 can clearly be seen, which caused a mean significant increase of 2 °C.

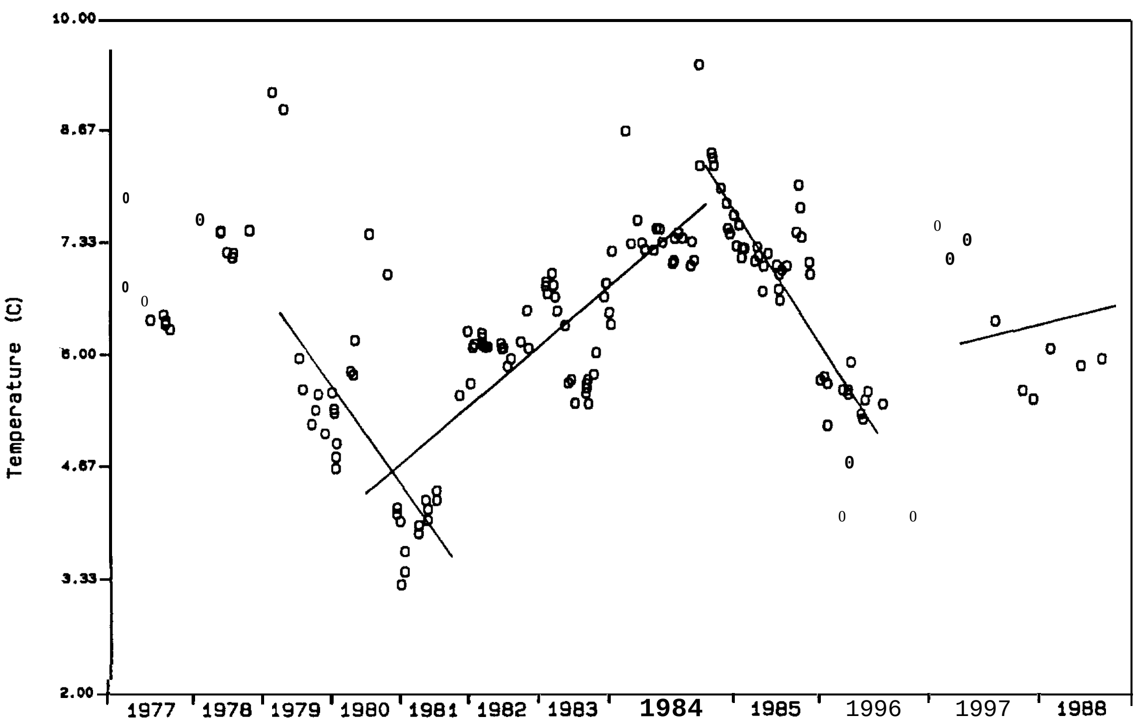


Figure 19. Temperature variation in 78-82 m water depth in the Bornholm Deep (BY 5 = BMP K2).

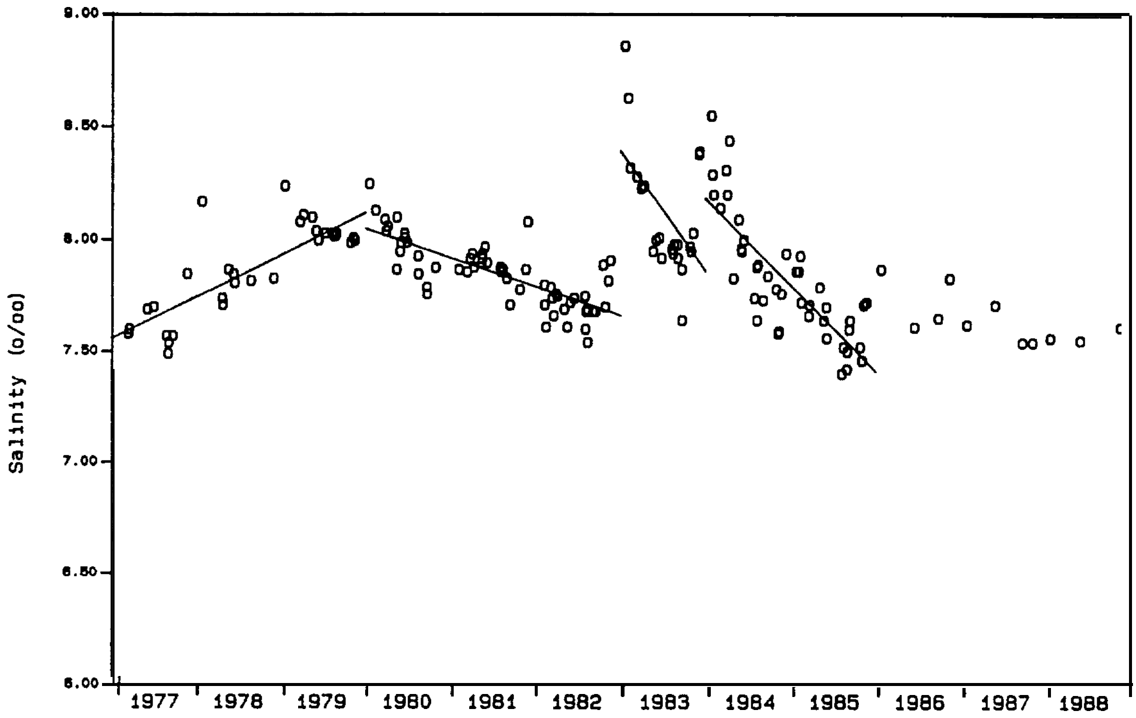


Figure 20. Salinity variation at the sea surface (0-2 m) in the Bornholm Deep (BY 5 = BMP K2).

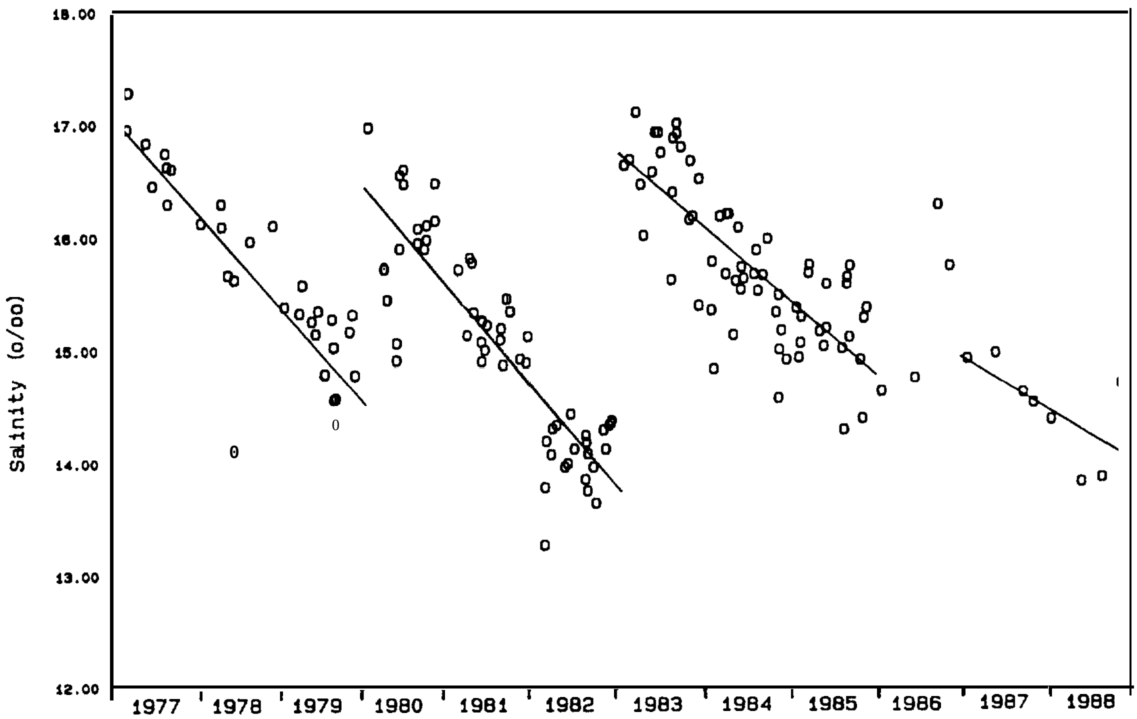


Figure 21. Salinity variation at 78-82 m depth in the Bornholm Deep (BY 5 = BMP K2).

The inflow of 198211983 is clearly seen in the salinity of the surface water. At BY 5 (BMP K2) the increase was about 1 PSU, see Fig. 20. After that the surface salinity decreases as a general trend. The decrease was strongest in the period 1983-1985, followed by a moderate decrease for the rest of the assessment period. One exception was late in 1983, when the salinity for a short period increased by about 0.5 PSU, which is larger than the mean annual variation of 0.2 PSU (Matthäus 1978). This increase cannot be explained by identified inflows to the Baltic.

The inflow of 198211983 caused an increase of almost 4 PSU in the deep water as is clearly demonstrated in Fig. 21. From 1983 to 1988 the trend is then decreasing in general. One exception was caused by the inflow of 1986, which caused a sharp increase of the salinity by almost 2 PSU, but only for a short period.

1.2.3.3 Eastern Gotland Basin, Gdaiisk Deep and the eastern area of the Northern Baltic Proper

W. Matthäus[2], J. Elken[6] and B. Cyberska[5]

The variations in the Eastern Gotland Basin are demonstrated with data of the representative station Gotland Deep BY 15 (BMP 11) and supported by data from the stations P 1 (BMP L1; Gdansk Deep) and BY 28 (BMP H2; Northern Baltic Proper).

Table 5. Characteristic changes in the Gotland Deep (BY 15 = BMP 11) during the period 1977-1988 (after Matthäus 1990)

Parameter	Depth	Overall trend	Annual changes
Salinity	O m	-0.5 PSU	-0.04 PSU/a
	100 m	-1.7 PSU	-0.15 PSU/a
	200 m	-1.1 PSU	-0.09 PSU/a
Temperature	100 m	-1.6 °C	-0.14 °C/a
	200 m	-1.4 °C	-0.12 °C/a
In situ density	100 m	-1.3 u-units	-0.11 u-units/a
	200 m	-0.7 u-units	-0.06 a-units/a
Depth of isohaline	8 PSU	-20.2 m	-1.7 m/a
	10 PSU	-28.7 m	-2.5 m/a
	12 PSU 1)	-95.9 m	-8.8 m/a
Depth of halocline		-9.6 m	-0.8 m/a

1) 1977 - Jan 1988

The three severe winters 198411985 - 198611987 caused partly considerable negative temperature anomalies between -0.5 °C and more than -1 °C in the surface water of the Eastern Gotland Basin. The negative anomalies were conserved in the intermediate water layer until autumn (-1 °C to -3 °C, cf. Nehring and Francke 1987a, b, 1988). Because of the absence of major Baltic inflows the surface salinity, on average, decreased by -0.5 PSU (cf. Table 5). The inflows of higher saline water between November 1982 and January 1983, in spring 1986 and in fall 1988 had hardly effects on the surface salinity (cf. Figs. 23 and 24).

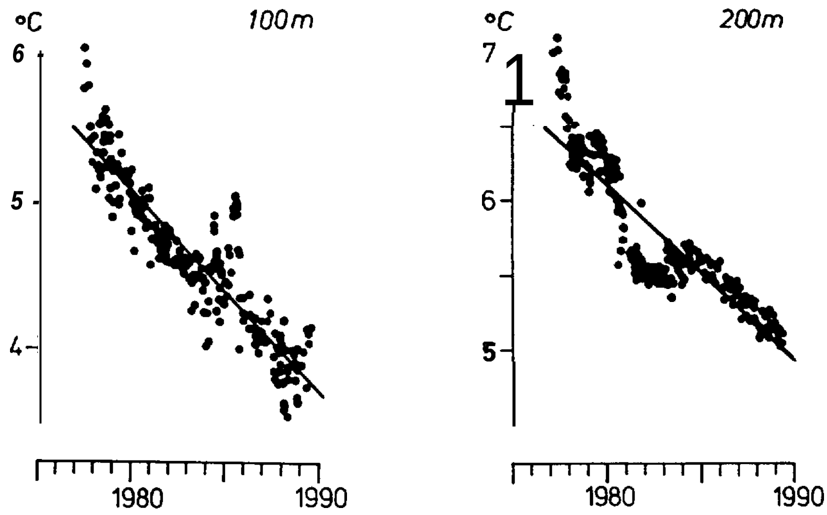


Figure 22. Variations of temperature in the Gotland Deep (BY 15 = BMP J1) during 1977-1989 at 100 and 200 m depth.

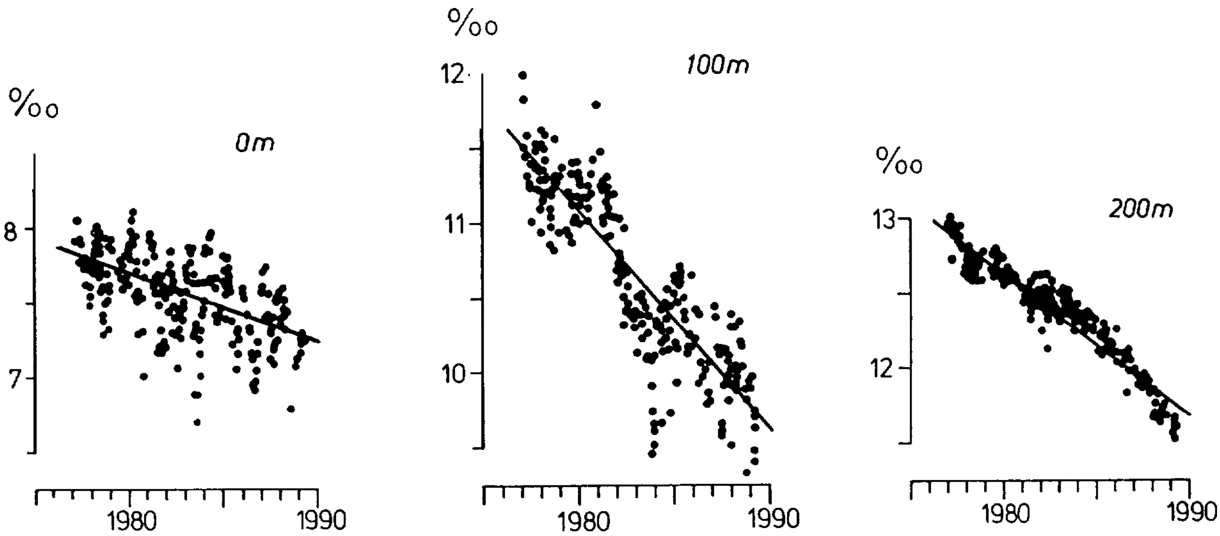


Figure 23. Variations of salinity in the Gotland Deep (BY 15 = BMP J1) during 1977-1989 at the surface and at 100 and 200 *m* depth.

The stagnation period in the Eastern Gotland Basin deep water, starting after major inflows at the turn of 1975/1976 and during late 1976, has continued since more than 13 years. The inflows 1982/1983, 1986 and 1988 had only minor effects on the conditions in the deep water.

Clear indications of the inflowing water bodies are only in the Gdaiisk Basin (cf. Figs. 24-25). There is, however, a distinct mean decrease in salinity and temperature of the deep water in the Gdaiisk Basin (cf. Table 6).

The inflowing water passed the Eastern Gotland Basin immediately below the halocline in the layer between 80 and 125 m (cf. Figs. 22-23).

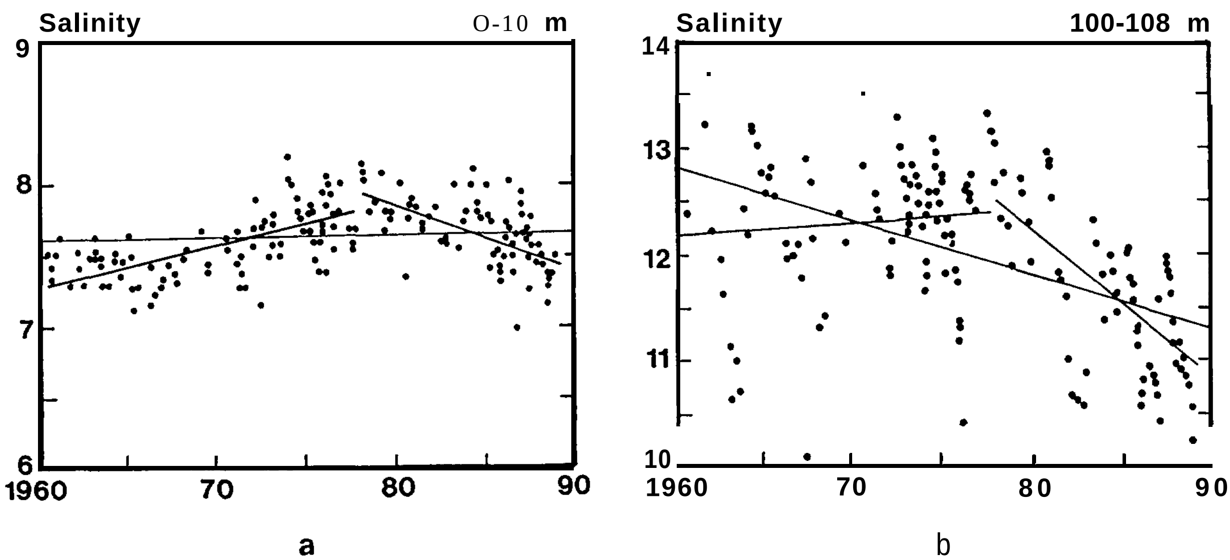


Figure 24. Long-term variations in salinity (PSU) of the Gdańsk Deep waters (Station P 1 = BMP L1).
a. surface (O-10 m)
b. bottom water (100-108 m)

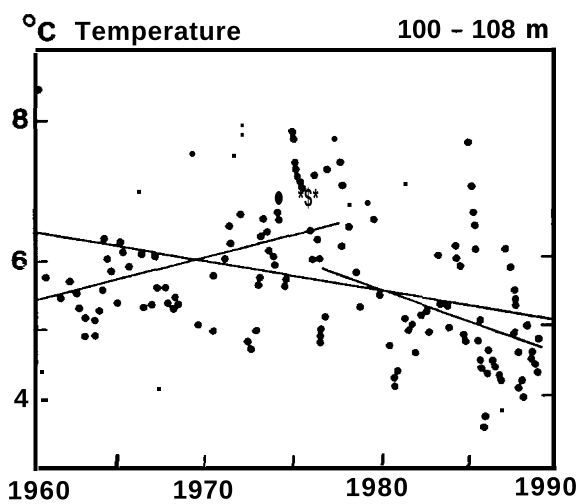


Figure 25. Long-term variations of the water temperature (°C) in the bottom water (100-108 m) in the Gdańsk Deep (Station P 1 = BMP L1).

Table 6. Characteristic changes in the Gdaiisk Deep (Station P1 = BMP L1) during the period 1978-1988.

Parameter	Depth	Overall trend	Annual changes
Salinity	0- 10 m	-0.5 PSU	-0.05 PSU/a
	80 m	-2.4 PSU	-0.21 PSU/a
	100-108 m	-1.7 PSU	-0.15 PSU/a
Temperature	80 m	-1.6 °C	-0.14 °C/a
	100-108m	-0.9 °C	-0.08 °C/a

During the stagnation period a decrease of salinity, temperature and density in the deep water of the Gotland Deep and a considerable shift in the isohalines and the halocline centre to greater depths occurred (cf. Figs. 22-23 and Table 5). The period started with the highest temperatures ever measured in the deep water (e.g. 7.1 °C in 200 m level of the Gotland Deep, BY 15 = BMP 11). The salinity and density values observed in the end of 1988 - 11.6 PSU and 10.2 u-units, respectively - were among the lowest measured since the 1930s at the 200 m level (Matthäus 1987). With reference to the depth of the halocline centre and the isohalines, the shallowest depths ever observed were found at the beginning of the stagnation period. The almost continuous shifting of the 10- and 11-PSU-isohalines down to more than 100 m and 140 m, respectively, finished at the greatest depths observed since the beginning of the century. Since January 1988, the salinity of the deep water has been permanently smaller than 12 PSU.

Considerable changes of the water mass structure in the halocline and the deep layers occurred by T-S relations in winter/spring of 1979/1980 (the Gotland Deep, station BY 15 = BMP J1, Fig. 26) and in summer/autumn of 1985 (the whole Eastern Gotland Basin, stations BY 15 (BMP J1) and BY 28 (BMP H2)). Then the water corresponding to the fixed isohalines (isopycnals) became colder by more than 1 °C in the upper part of the halocline but less in the deep layers. The 1985 cooling can be attributed to the anomalously cold winter of 198411985 (see Section 1.1) which caused increased downward cold water flux due to cross-isopycnal mixing. Thermal homogenization (decrease of temperature versus salinity gradient) took place from summer of 1983 to summer of 1985.

Until the end of 1988 the stagnation period led to a mean decrease by 1 PSU in salinity, by 1.4 °C in temperature and by 0.7 u-units in density at the 200 m level in the Gotland Deep (cf. Table 5). During the same period, the isohalines shifted to greater depths by 20 m (8 PSU) to 96 m (12 PSU).

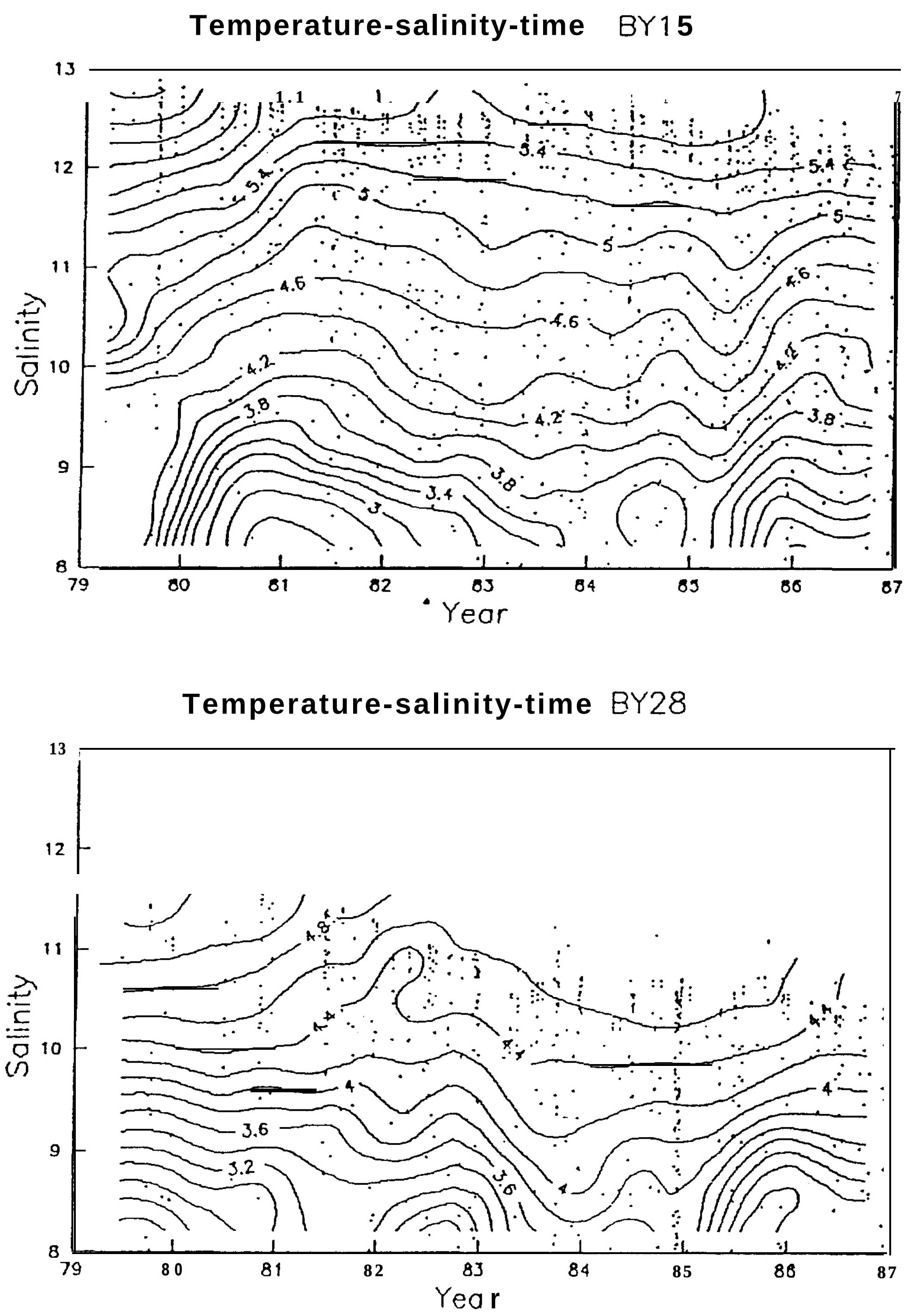


Figure 26. Temperature contour plots versus salinity and year from the data of stations BY 15 (BMP J1) and BY 28 (BMP H2).

Variation of spatially mean hydrographic parameters and their heterogeneity (standard deviation) in the Gotland Deep has been studied on the basis of the data of repeated CTD polygon surveys. The data set was restricted with the box of 57-58° N, 19-21° E with depths greater than 110 m which yielded altogether 403 CTD casts from 1984 to September 1988. Partition of the data by about 10 days interval gave 22 guasisynoptic spatial data subsets.

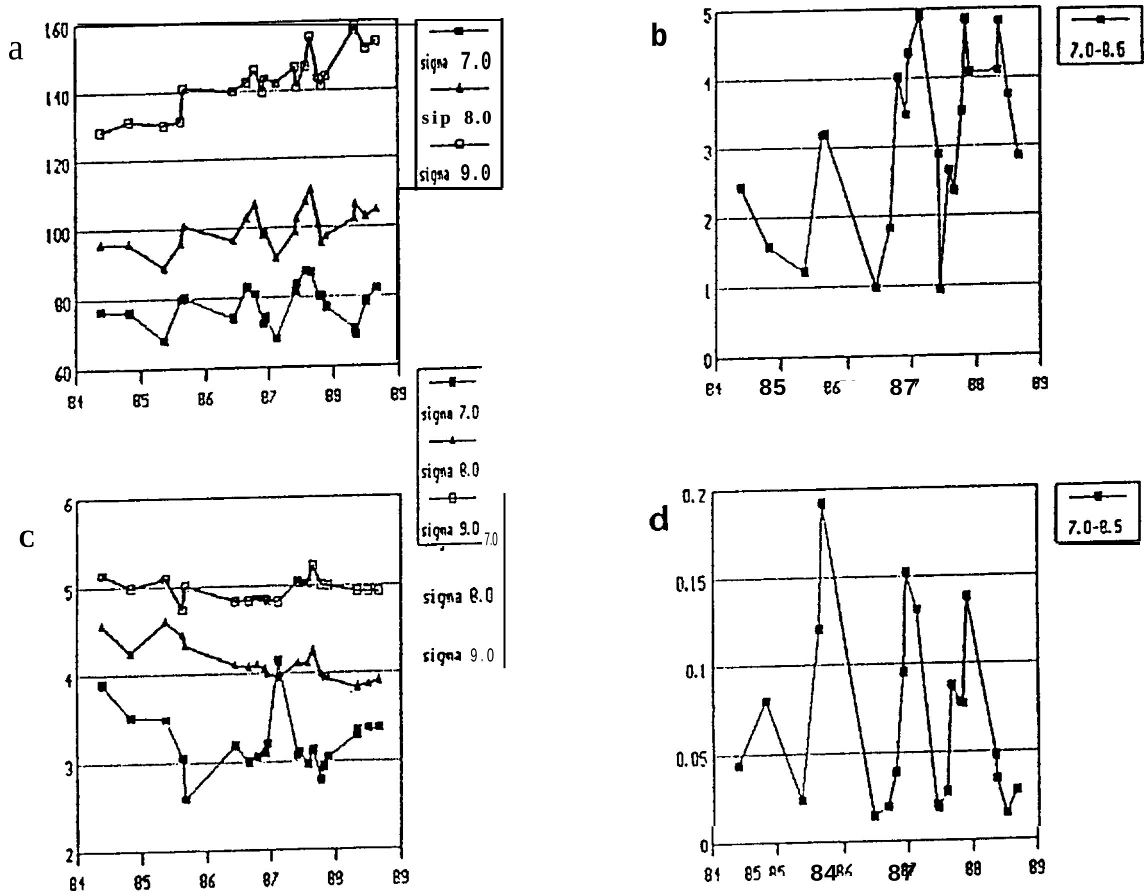


Figure 27. Spatially mean values of isopycnal depth (a) and temperature (c) and corresponding standard deviations averaged on isopycnals from 7 to 8.5 kg/m^3 (b and d) in the area 57-58° N; 19-21° E.

The stagnation period resulted in deepening of isopycnals and decrease of density stratification (Fig. 27a) for the halocline and the deep layers. Absence of the major inflows of the isopycnally warmer water caused overall decrease of temperature on isopycnals (Fig. 27c). However, a slight increase of temperature on isopycnals 8 and 9 kg/m^3 in spring/summer of 1987 and offset of the trends is related to the 1986 spring inflow. Spatial heterogeneity has remarkable seasonal variation, being higher in the winters and lower in the summers (Fig. 27b, d). Exception here was the summer of 1985 when abrupt temporal changes in T-S relation (Fig. 26) were accompanied by increased spatial heterogeneity of isopycnal temperature. The winter activization is explained by increased forcing of meso- and small-scale motion in case of the absence of the thermocline.

In view of these characteristics the current stagnation period must be regarded as the longest stagnation interval ever recorded during this century and caused extreme variations which never have been observed since the start of oceanological observations in the Baltic Sea.

1.2.3.4 Western Gotland Basin

S. Carlberg[4]

The hydrographical development in the Western Gotland Basin is demonstrated with data from the Karlsö Deep (BY 38 = BMP 11) and supported by data from the Landsort Deep (BY 31 = BMP H3) in the Northern Basin.

The temperature variability is high in the water layer down to at least 50 m. The span of the temperature variation in the assessment period 1984-1988 is almost identical to what was observed for the previous years after 1977. In the deeper layers, at 80 m and even at 100 m, a seasonal variation can clearly be seen, see Figures 28-29. However, this does not conceal the fact that there is a very clear trend of decreasing deep water temperatures during the whole period. In fact for the period plotted, 1977-1988, the temperature is decreasing continuously. The temperature conditions described also apply to various water masses of the Landsort Deep.

The salinity field in the Western Gotland Basin is in all water depths dominated by a decreasing trend; not only in the assessment period but all since 1978, see Figures 29 and 30. Due to the time delay caused by advection, and the smoothing effect caused by gradual mixing of water masses, the salt water inflows identified in the Arkona and Bornholm Basins are not easily discernable at BY 31 (BMP H3) and BY 38 (BMP 11). If the inflows are used for reference the increase in salinity at BY 38 occurs with different time delay after each inflow, see Figures 30 and 31 a-b. The same conditions are obvious in the deep water of the Landsort Deep (BY 31) as described in Figures 32-33; the short-term variability is smaller and thus the long-term decrease of salinity is more obvious.

The changes of temperature and salinity are summarized in Table 7.

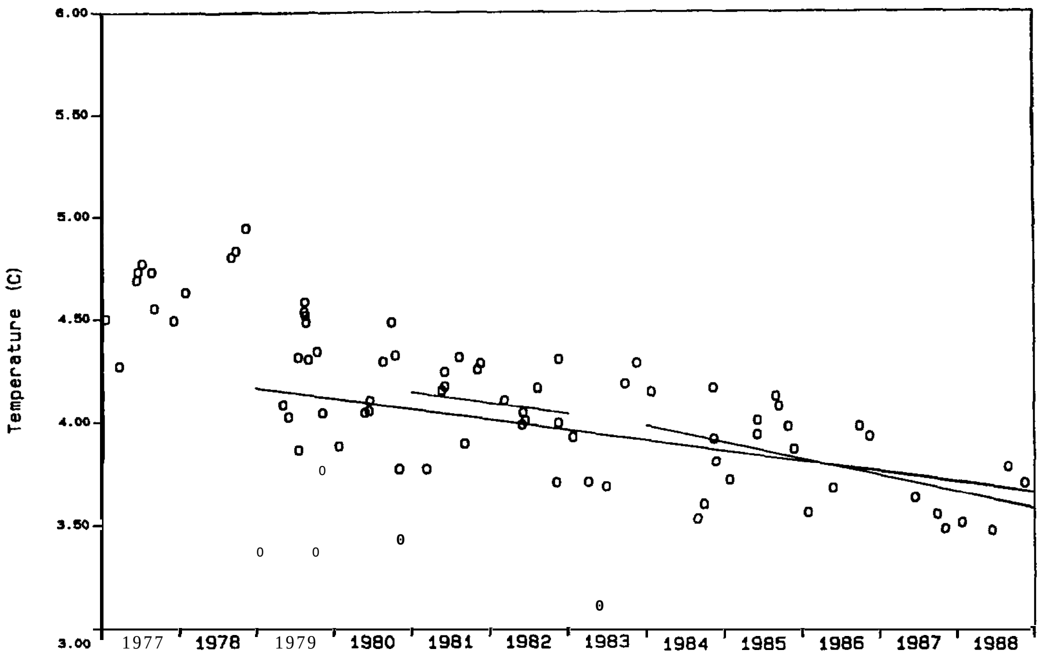


Figure 28. Temperature variation at the Karlsö Deep (BY 38 = BMP I1), 78-82 m.

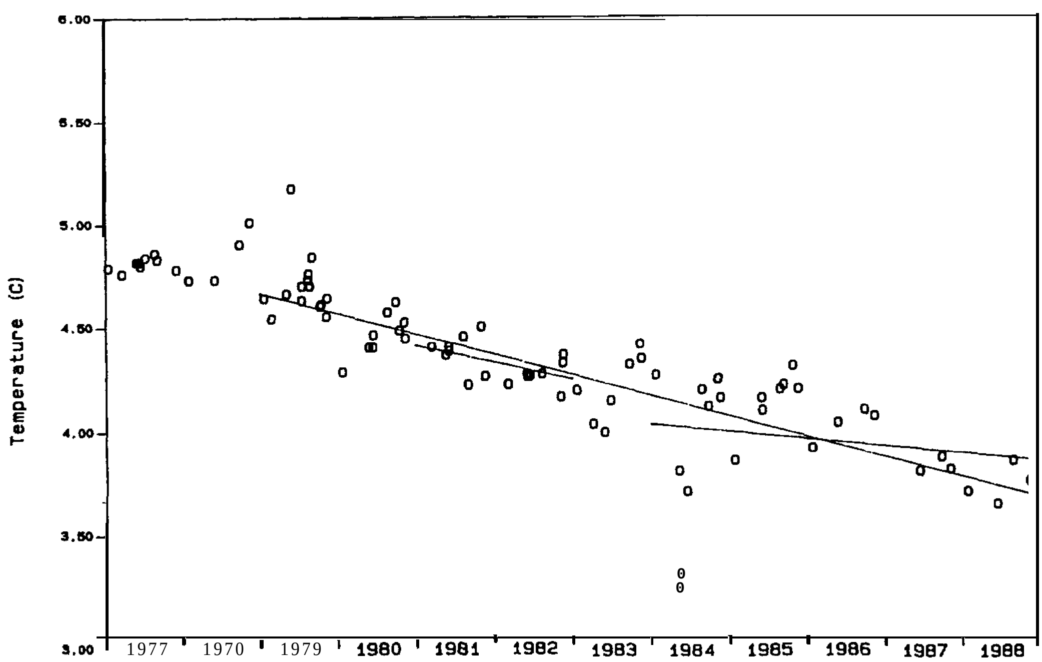


Figure 29. Temperature variation at the Karlsö Deep (BY 38 = BMP I1), 95-105 m.

Table 7. Characteristic changes of temperature and salinity in the Western Gotland Basin at the stations Landsort Deep, BY 31 (= BMP H3) and Karlsij Deep, BY 38 (BMP 11).

Area	Para-meter	Period	Depth	Overall trend	Annual changes
Landsort Deep	S	1979-1984	190-210 m	-0.92 PSU	-0.15 PSU/a
Landsort Deep	S	1985-1988	190-210 m	-0.22 PSU	-0.06 PSU/a
Landsort Deep	S	1978-1988	390-410 m	-1.32 PSU	-0.12 PSU/a
Landsort Deep	S	1980-1984	390-410 m	-0.80 PSU	-0.16 PSU/a
Landsort Deep	S	1985-1988	390-410 m	-0.24 PSU	-0.06 PSU/a
Karlsö Deep	S	1979-1988	48- 52 m	-0.34 PSU	-0.03 PSU/a
Karlsij Deep	S	1984-1988	48- 52 m	-0.52 PSU	-0.10 PSU/a
Karlsö Deep	T	1979-1988	78- 82 m	-0.53 °C	-0.05 °C/a
Karlsij Deep	T	1981-1982	78- 82 m	-0.11 °C	-0.06 °C/a
Karlsij Deep	T	1984-1988	78- 82 m	-0.41 °C	-0.08 °C/a
Karlsij Deep	T	1979-1988	95-105 m	-1.00 °C	-0.10 °C/a
Karlsö Deep	T	1981-1982	95-105 m	-0.17 °C	-0.09 °C/a
Karlsij Deep	T	1984-1988	95-105 m	-0.20 °C	-0.04 °C/a

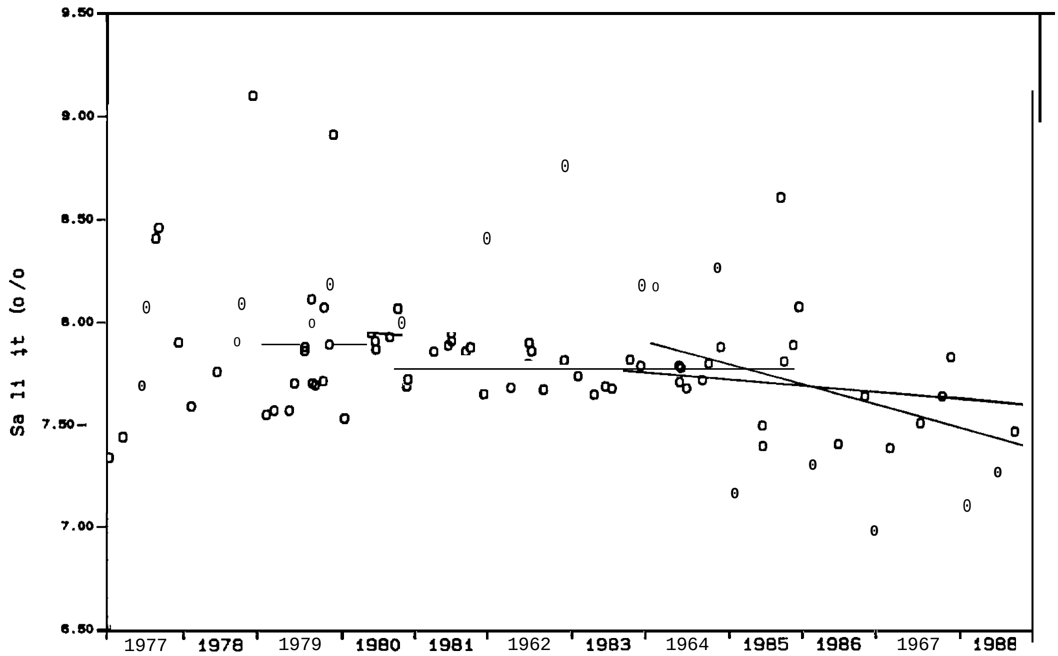
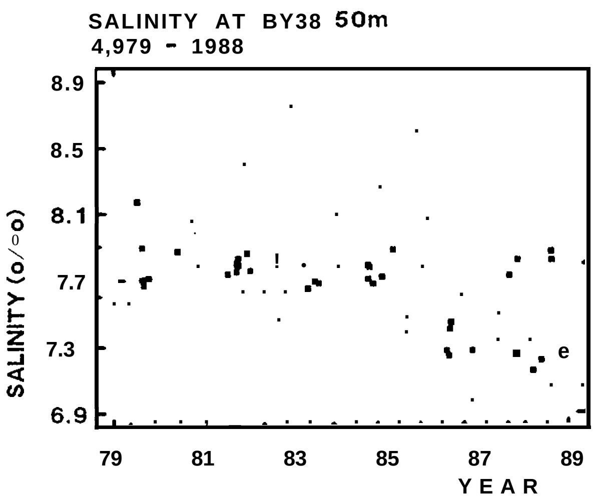


Figure 30. Salinity variation at the Karlsö Deep (BY 38 = BMP I1), 48-52 m.



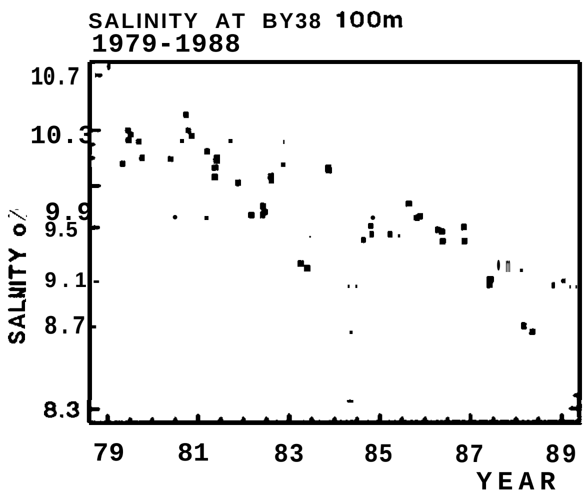


Figure 31 a. Salinity variation at the Karlsö Deep (BY 38 = BMP I1), 50 m (BMP data only).

b. Salinity variation at the Karlsö Deep (BY 38 = BMP 11), 100 m (BMP data only).

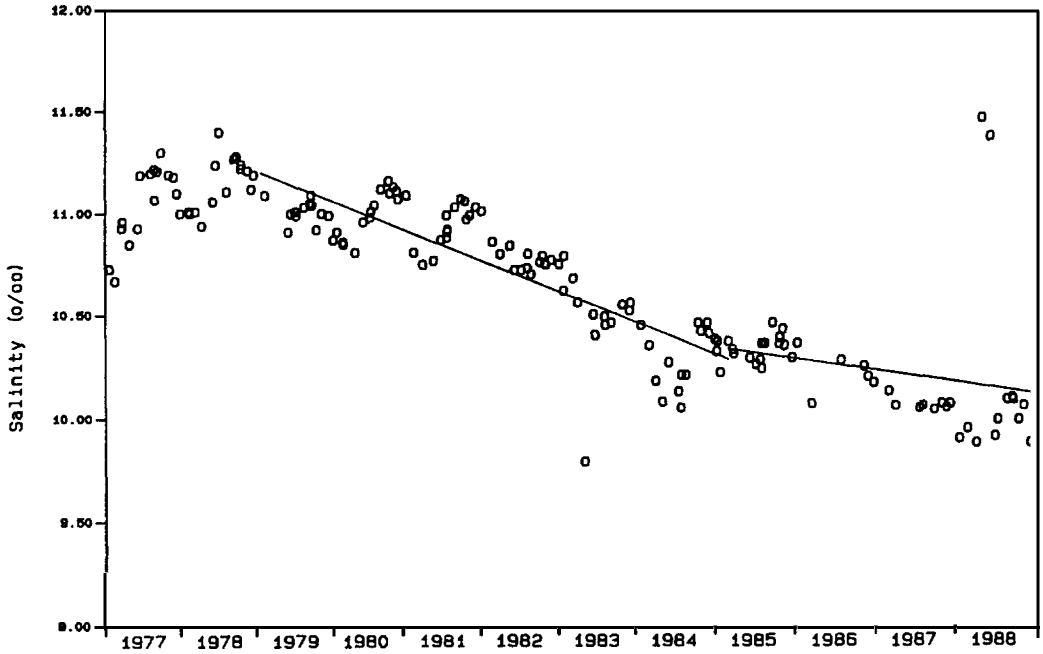


Figure 32. Salinity variation at the Landsort Deep (BY 31 = BMP H3), 190-210 m.

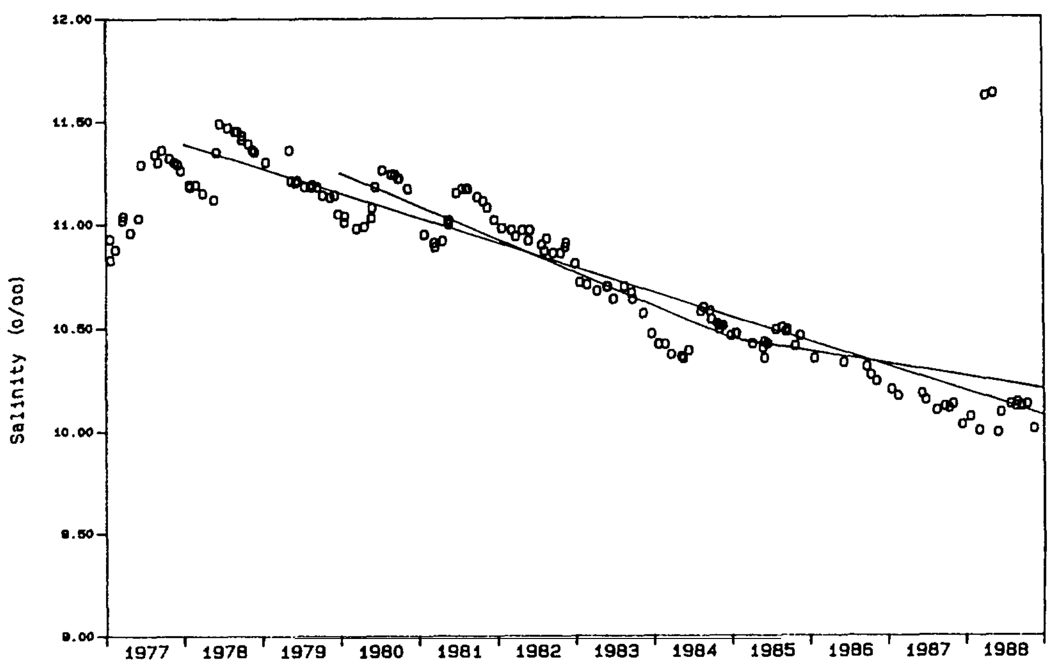


Figure 33. Salinity variation at the Landsort Deep (BY 31 = BMP H3), 390-410 m.

1.2.4 Gulf of Finland

V. Astok[1], R. **Tamsalu**[6], A. **Nõmm**[1] and Y. **Suursaar**[6]

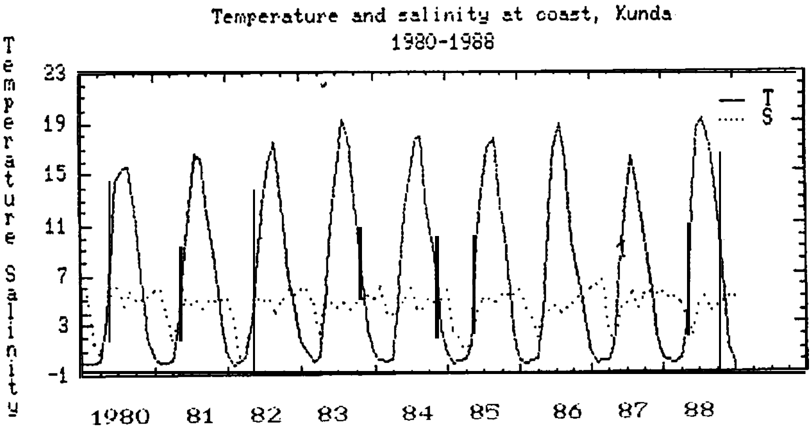
The hydrographical development in the Gulf of Finland during the 1980s is demonstrated using HELCOM data from the BMP data bank and Estonian coastal stations data concerning surface layer as well as those collected by the Estonian Hydrometeorological Service research vessels within the national monitoring system.

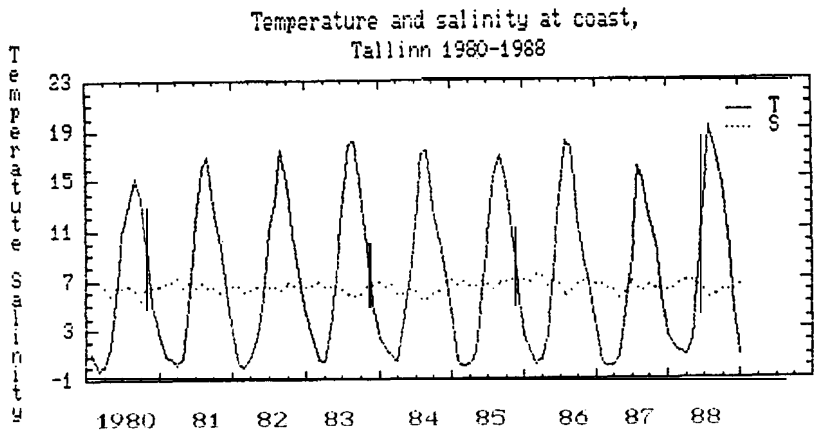
The temperature in the surface layer (Fig. 34) shows normal annual variation in all three presented stations (Kunda is located ca 100 km east, Heltermaa ca 100 km west from Tallinn). The cold summers of 1985 and 1987 were interchanged by warmer 1984 and 1986; the summer 1988 was extremely warm. In the open part of the Gulf of Finland the variation is very strong (Fig. 35) and the calculated trend for temperature cannot be taken into consideration. However, the trend *in* the bottom layers (80 ... 100 m) temperature seems to be confidential and is in good correlation with the decrease in salinity (Fig. 36). The value of the temperature trend in bottom layer is -1,7°C per 20 years or -0.085°C/a.

The salinity in the surface layer shows a clear variation in the annual cycle, especially in the eastern part of the Gulf of Finland (Fig. 34). Trend estimation from open sea observations gives -0,02 PSU/a when using data obtained in Estonia during 1969-1988 (Fig. 37). The same estimated value from BMP data for the last ten years (station LL7=F3) is -0,04 PSU/a (Fig. 39).

In the bottom layer the decreasing of salinity is shown more clearly (Figs. 38 and 39). The trend values are -0,08 PSU/a (Estonian data, 20 years) and -0,11 PSU/a (BMP data, 10 years).

It can be concluded that during the last 20 years there have been no large inflows of the saline, but oxygen-poor deep waters from the Baltic Proper into the Gulf of Finland. The conclusion is confirmed by the increasing O_2 concentrations in the deep layers of the Gulf of Finland (Fig. 40).





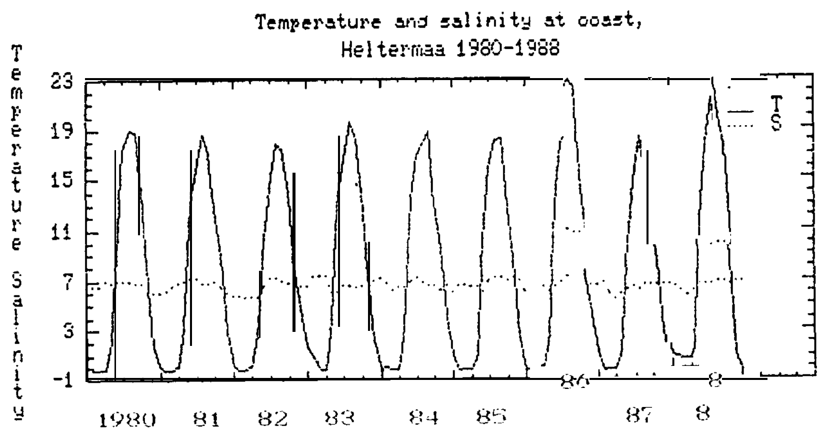


Figure 34. The surface water temperature and salinity (daily measurements, monthly means) at the Estonian coastal stations 1980-1988.

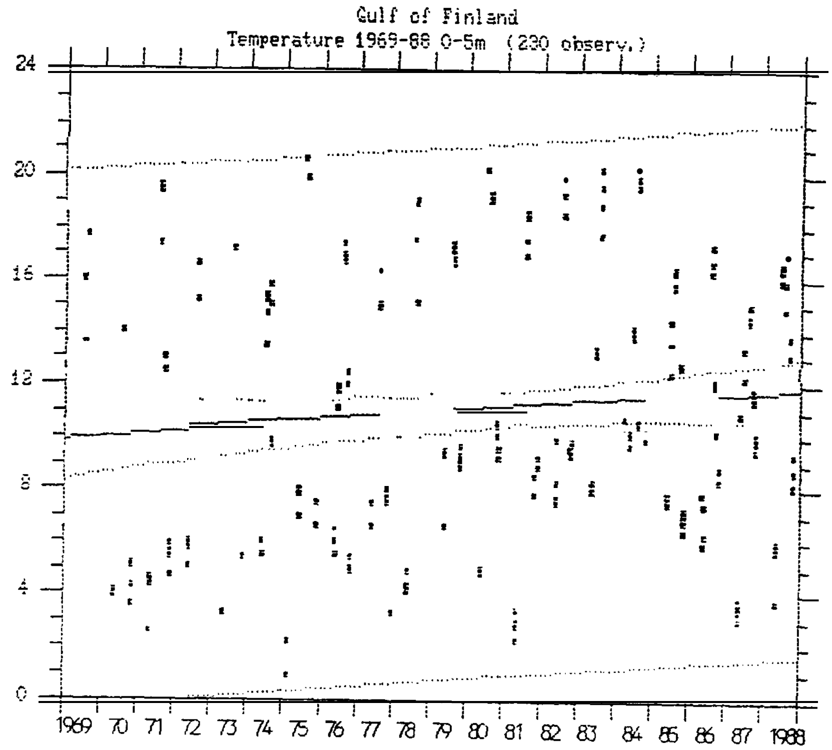


Figure 35. Surface water temperature variation 1969-1988 in the Gulf of Finland 1969-1988.

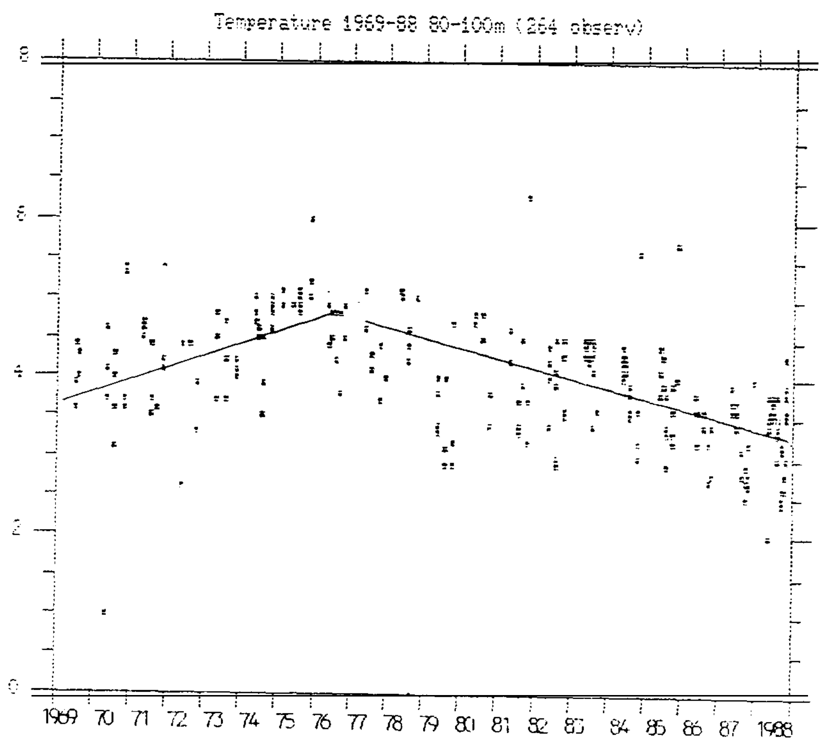


Figure 36. Water temperature variation 1969-1988 in the bottom layer (So-100 m) in the Gulf of Finland.

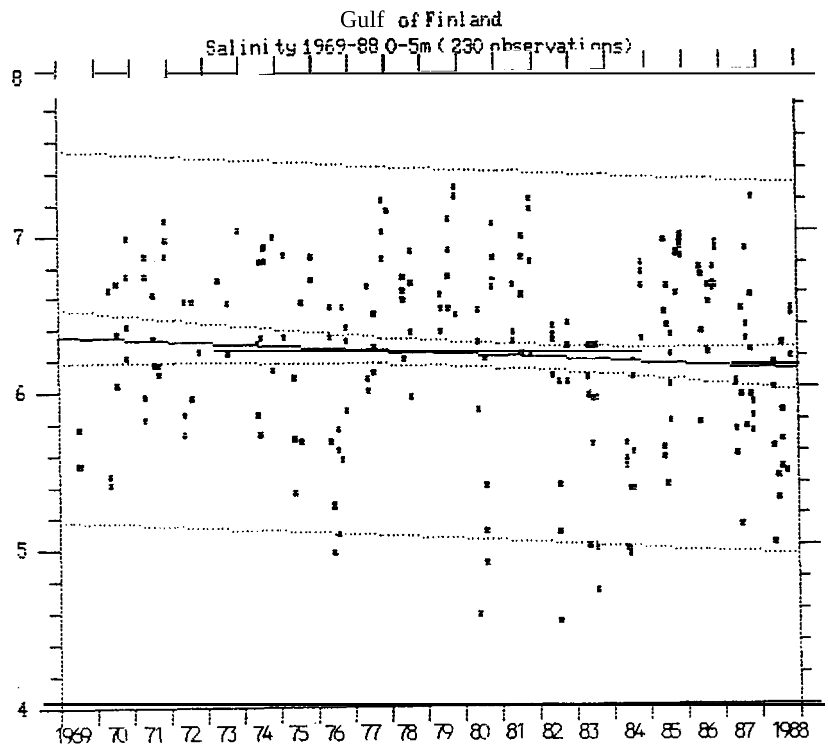


Figure 37. Surface salinity variation 1969-1988 in the Gulf of Finland.

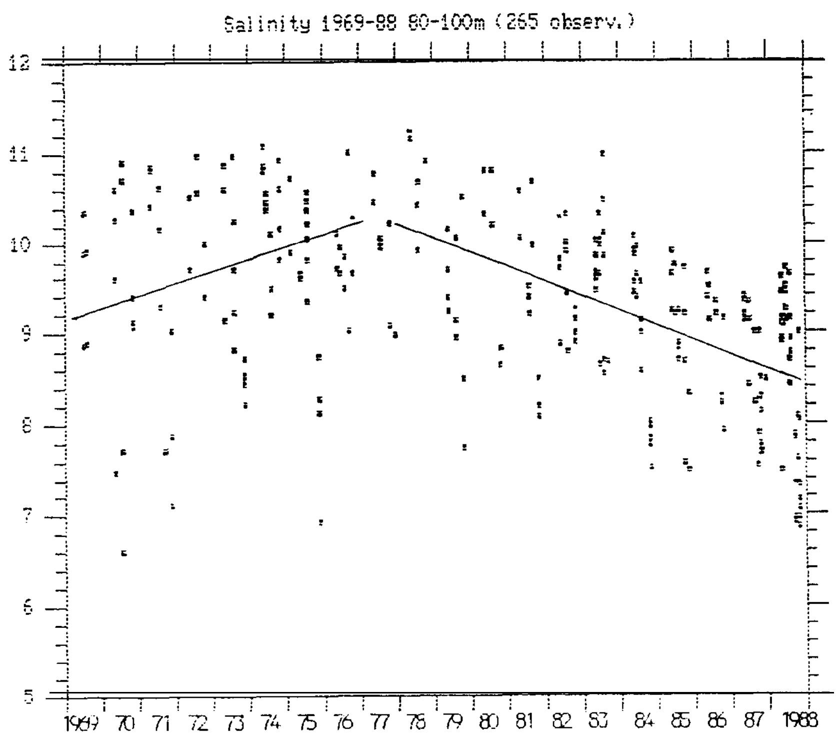


Figure 38. Salinity variation 1969-1988 in the bottom layer (80-100 m) in the Gulf of Finland.

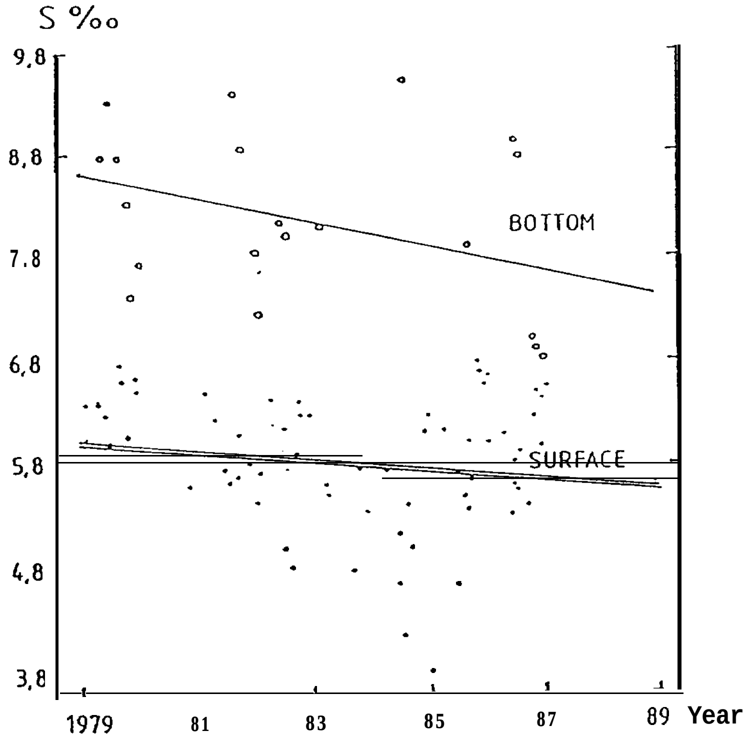


Figure 39. Salinity variation 1979-1987 in the bottom and surface layers of the Gulf of Finland (station LL7 = F3).

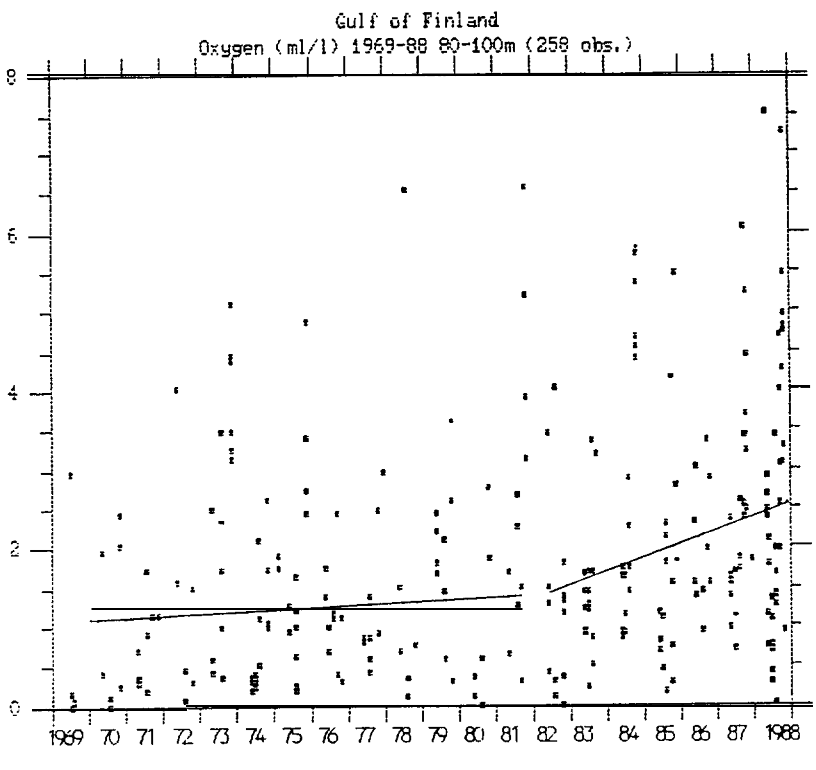


Figure 40. Variation of oxygen concentration 1969-1988 in the bottom layer (SO-100 m) in the Gulf of Finland.

1.2.5 Gulf of Riga

V. Astok[1] and V. Berzins[3]

The hydrographical development in the Gulf of Riga during the last decades is demonstrated using averaged annual data from the Latvian national monitoring system (13 stations).

The annual mean air and water temperature (Fig. 41) shows large year-to-year variations; the warm years (1984, 1986, 1988) are warmer than in the Baltic Proper, the cold years (1985, 1987) - colder, respectively. This is caused by the nearness of land to the practically enclosed gulf; the climate in the region of the Gulf of Riga is more continental than in the other parts of the Baltic Sea area.

The long-term variations in the river run-off and mean salinity are demonstrated in Figure 42. Minimum run-off in 1984 is accompanied by increasing salinity; however, in 1985 the salinity was higher though the river run-off was nearly 50% larger than in 1984. The changes in the mean salinity depend not only on fresh water run-off, but also on saline water inflows through the Strait of Irben. These inflows during 1984 and 1985 were caused by suitable meteorological conditions in this region. The aforementioned increase in salinity, which is of local importance, differs from values obtained in the other parts of the Baltic Sea.

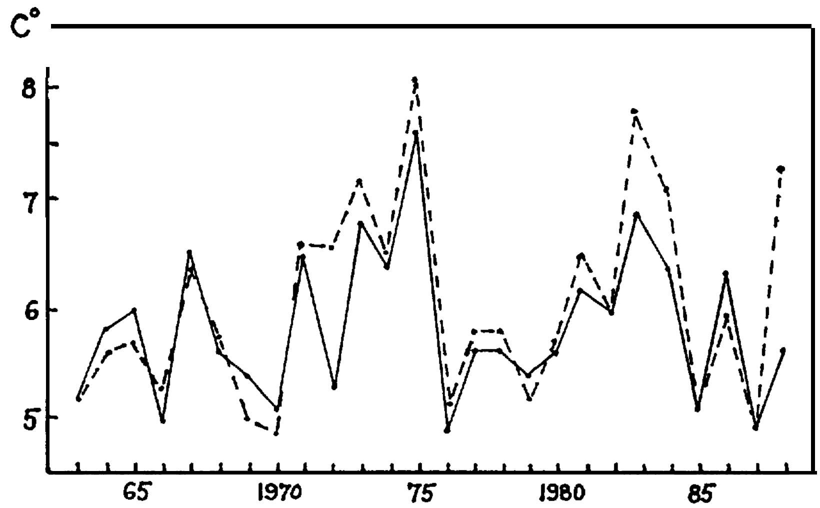


Figure 41. The mean air (_) and water (_ _ _) temperature of the Gulf of Riga.

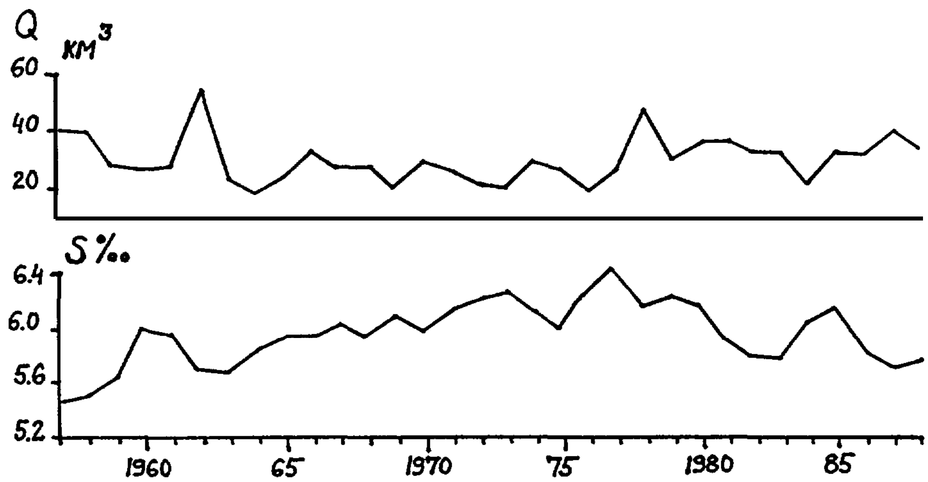


Figure 42. The river run-off (Q) and mean salinity (S) of the Gulf of Riga.

1.2.6 Gulf of Bothnia
S. Carlberg[4]

The hydrographical development in the Gulf of Bothnia is demonstrated with data from the representative stations SR 5 (BMP C4) in the Bothnian Sea and BO 3 (BMP A3) in the Bothnian Bay (Fig. 43.).

The unusually cold winters of 1984/1985 and 198611987 did not produce any pronounced effects in the surface layers in the Gulf of Bothnia. The temperature minima observed were not lower than those in most of the years since 1979. Down to about 50 m there is no long-term temperature trend for the assessment period. However, at 100 m and below the temperature at SR 5 (BMP C4) is clearly decreasing from 1984 to the middle of 1988 (Figure 44). This can also be seen at B03 (BMP A3) in Figure 45, although the lower number of observations there makes the picture less clear.' Figure 45 also demonstrates that in the bottom water of BO 3 the winter temperatures frequently approach 0°C, whereas at SR 5 the bottom water rarely is colder than about 1.6°C (Fig. 44).

Compared to the situation in the Baltic Proper the salinities are rather stable in the Gulf of Bothnia. In the surface water and at 50 m the variations are small and the trend in the Bothnian Sea may be described as weakly decreasing, see Figure 46. In the deep water at about 100 m and below this trend is more obvious as is shown in Figure 47. The decrease started in 1977-1979 at various depths and continued until late 1985 when the salinity suddenly increased by about 0.5 PSU. After that the decrease lasted until the end of 1988. The conditions in the Bothnian Bay are rather similar, although the greater scatter of the observations makes the trends less obvious. It is quite clear from Figure 48, however, that the salinity at 50 m shows a decrease during the period 1984-1988.

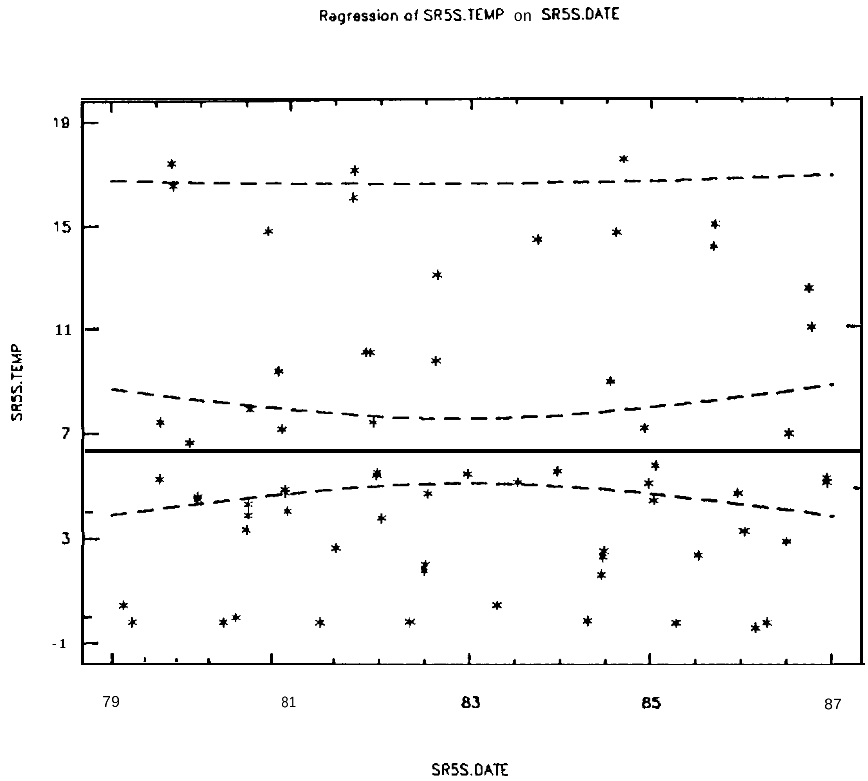


Figure 43. Temperature variation at sea surface of the station SR 5 (BMP C4) in the Bothnian Sea.

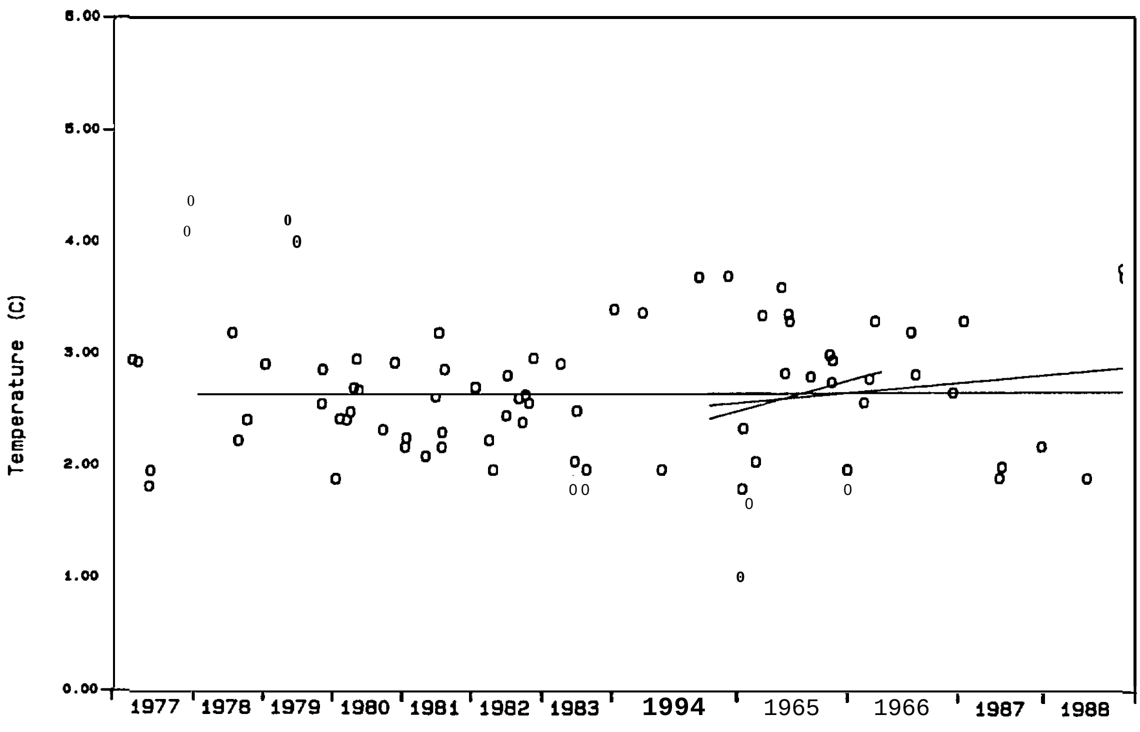


Figure 44 a. Temperature variation at the station SR 5 (BMP C4), 95-105 m in the Bothnian Sea.

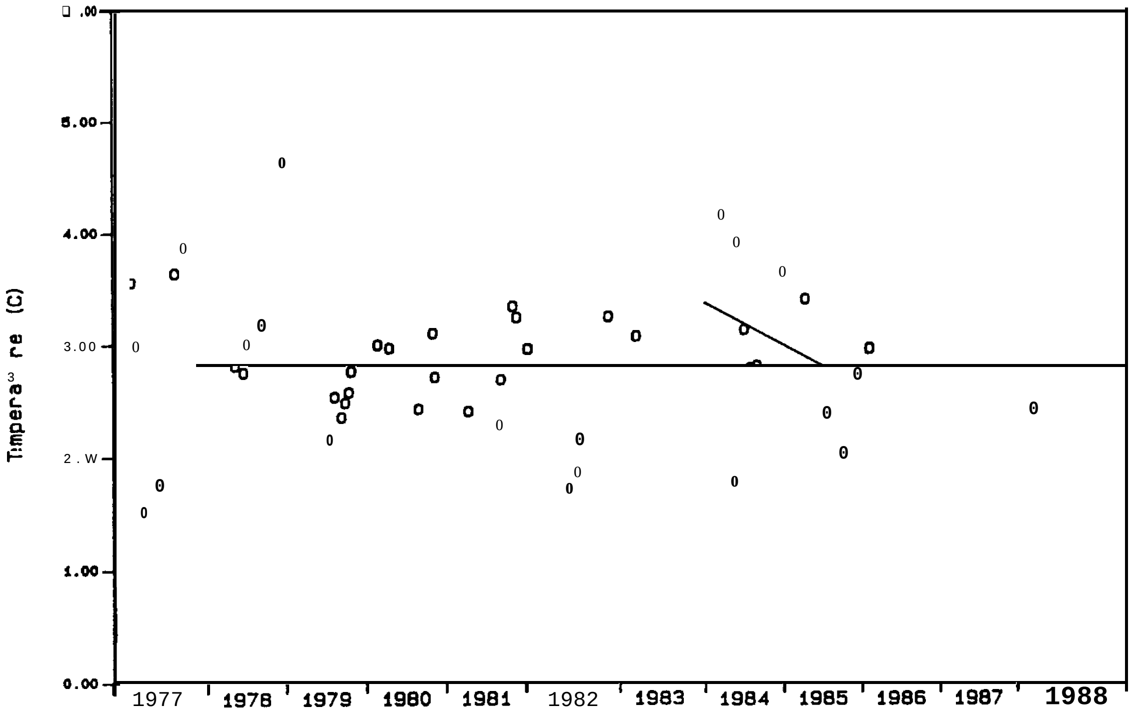


Figure 44 b. Temperature variation near bottom (115 m to bottom) at the station SR 5 (BMP C4) in the Bothnian Sea.

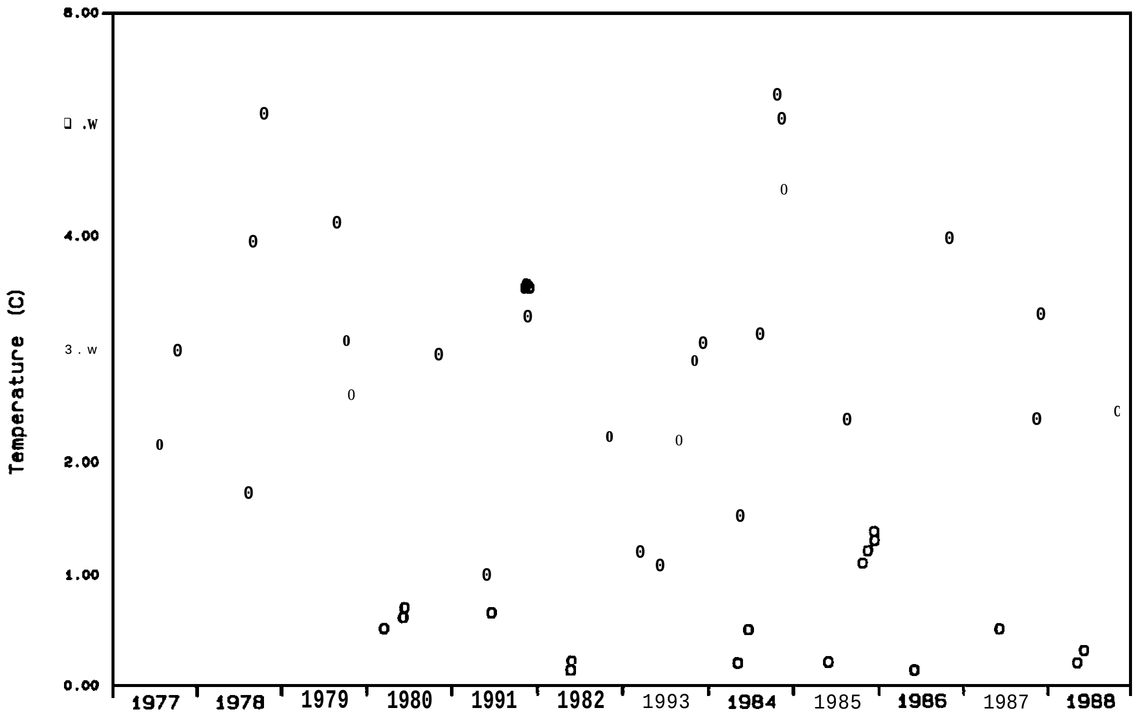


Figure 45. Temperature variation near bottom (95 m to bottom) at the station BO 3 (BMP A3) in the Bothnian Bay.

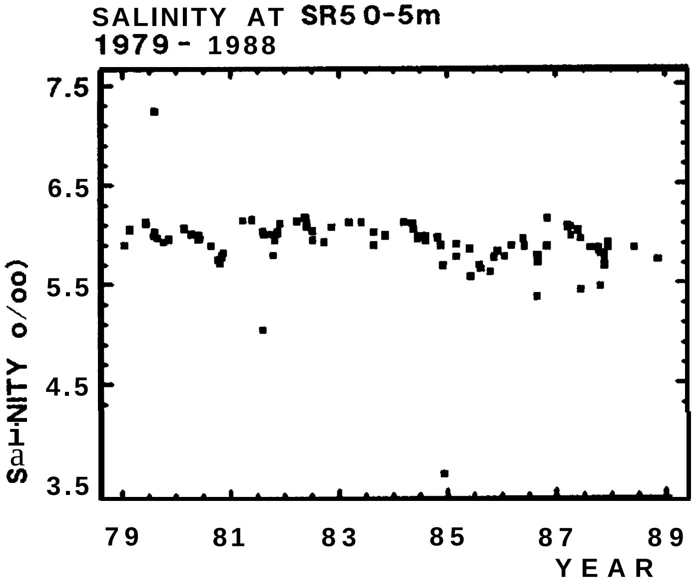


Figure 46 a. Salinity variation at sea surface of the station SR 5 (BMP C4) in the Bothnian Sea.

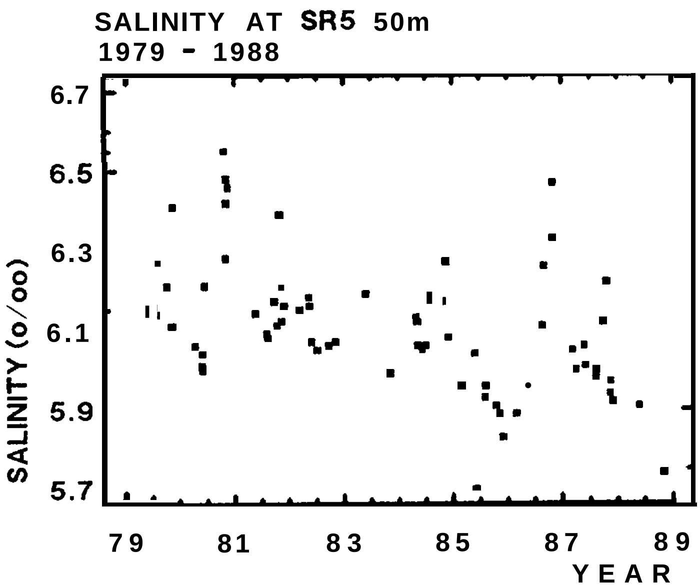


Figure 46 b. Salinity variation at the station SR 5 (BMP C4), 48-52 m.

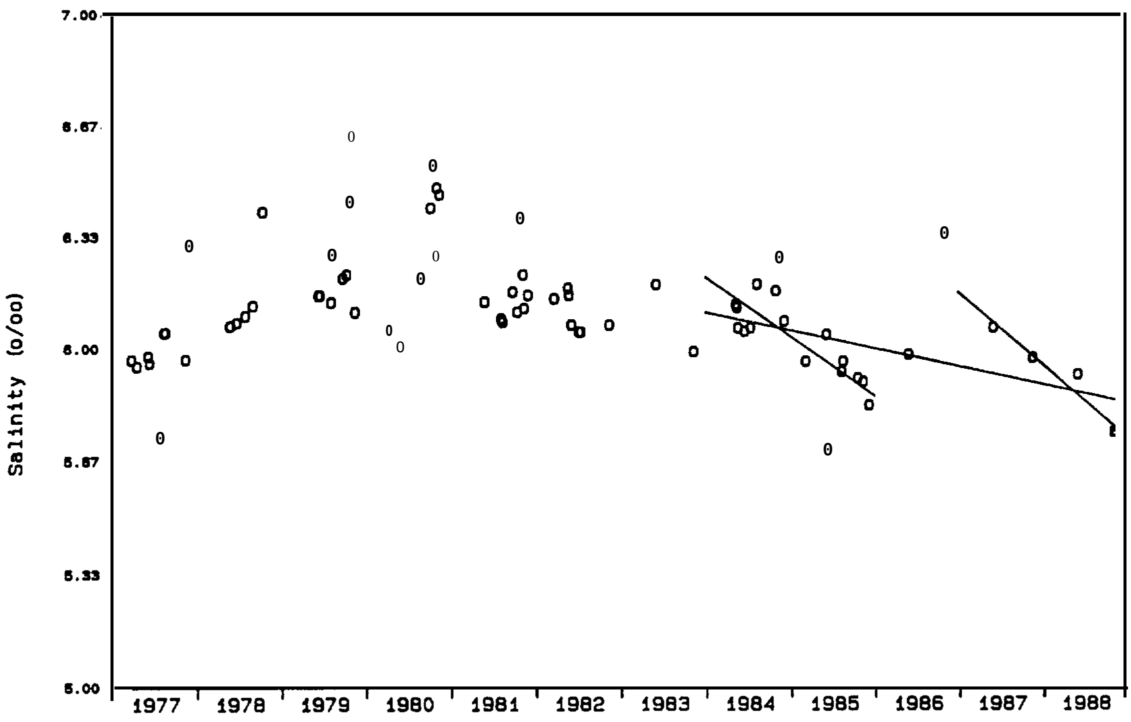


Figure 46 c. Salinity variation at the station SR **5 (BMP C4), 48-52** m. (drawn by SMHI for better evaluation of trends)

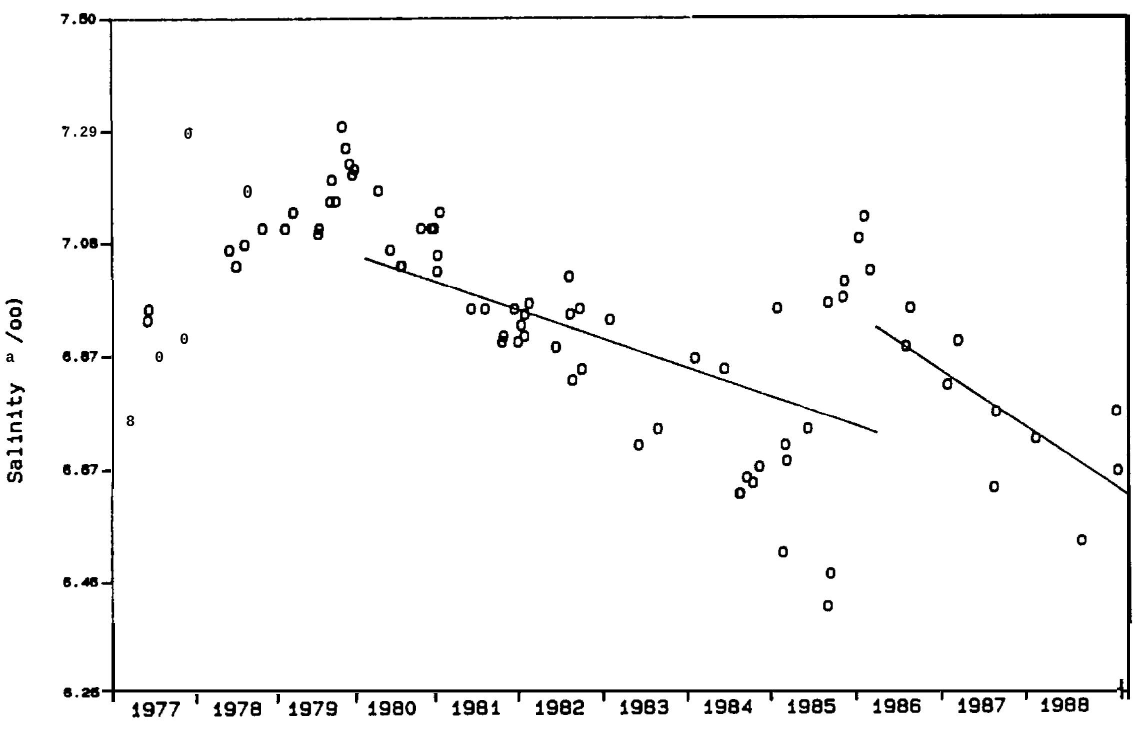


Figure 47 a. Salinity variation at the station SR 5 (BMP C4), 95-105 m.

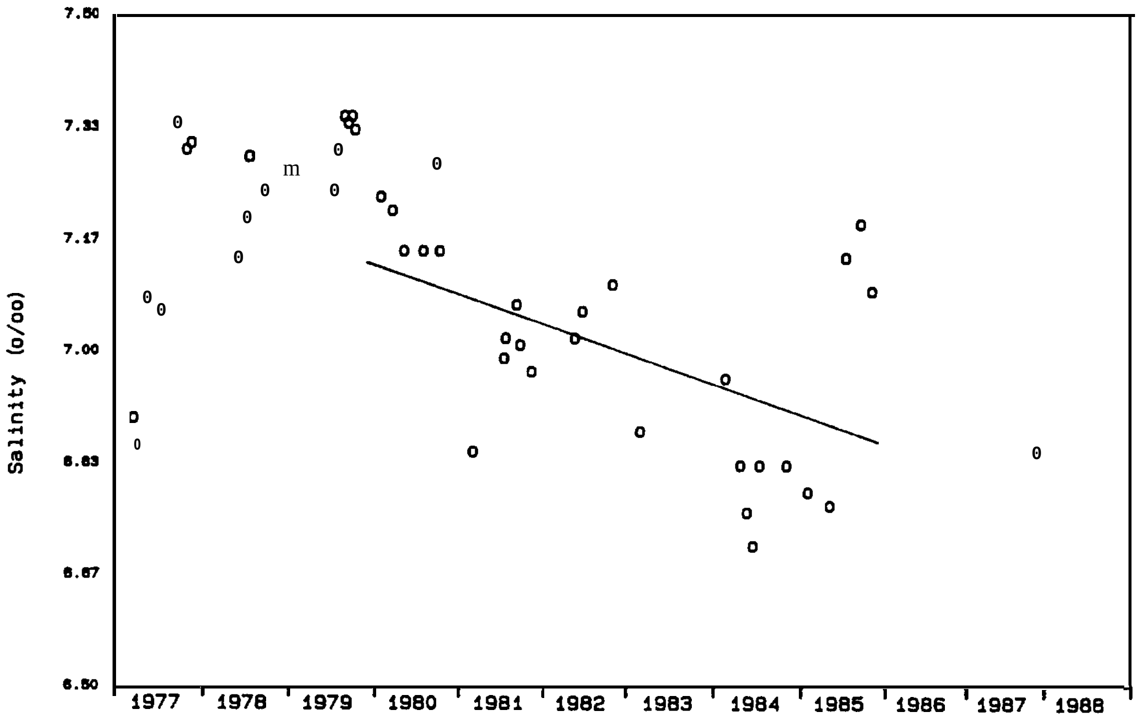


Figure 47 b. Salinity variation near bottom (115 m to bottom) at the station SR 5 (BMP C4).

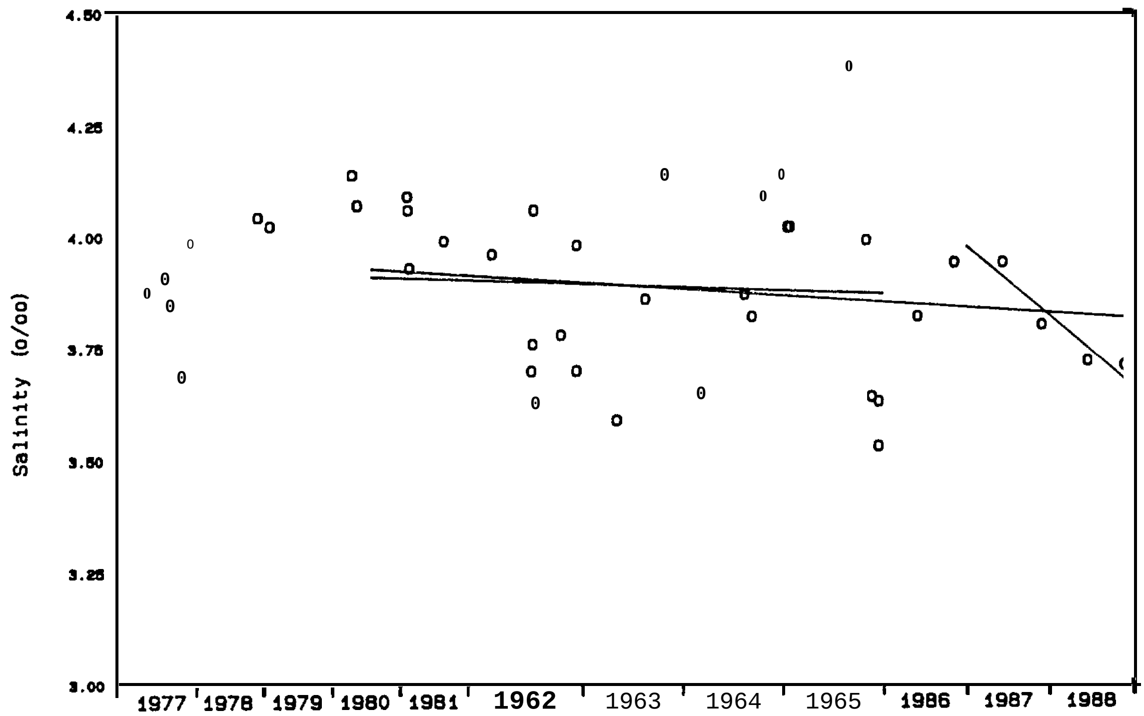


Figure 48. Salinity variation at the station BO 3 (BMP A3), 48-52 m.

The changes of temperature and salinity are summarized in Table 8 below.

Table 8. Characteristic changes of temperature and salinity in the Bothnian Sea (Station SR 5 = BMP C4) and the Bothnian Bay (Station BO 3 = BMP A3).

Area (Station)	Para-meter	Period	Depth	Overall trend	Annual changes
Bothnian Sea (SR 5 = BMP C4)	S	1984-1985	48- 52 m	-0.35 PSU	-0.18 PSU/a
Bothnian Sea	S	1987-1988	48- 52 m	-0.43 PSU	-0.21 PSU/a
Bothnian Sea	S	1984-1988	48- 52 m	-0.27 PSIJ	-0.05 PSU/a
Bothnian Sea	S	1980-1985	95-105 m	-0.33 PSU	-0.05 PSU/a
Bothnian Sea	S	1986-1988	95-105 m	-0.31 PSU	-0.10 PSU/a
Bothnian Sea	S	1980-1985	115 m-bottom	-0.55 PSU	-0.09 PSU/a
Bothnian Bay (BO 3 = BMP A3)	S	1980-1985	48- 52 m	-0.04 PSU	———
Bothnian Bay	S	1980-1988	48- 52 m	-0.12 PSU	-0.01 PSU/a
Bothnian Bay	S	1987-1988	48- 52 m	-0.32 PSU	-0.16 PSU/a
Bothnian Sea (SR 5 = BHP C4)	T	1978-1988	95-105 m	+0.06 °C	------
Bothnian Sea	T	1984-1985	95-105 m	+0.42 °C	+0.21 °C/a
Bothnian Sea	T	1984-1988	95-105 m	+0.34 °C	+0.17 °C/a
Bothnian Sea	T	1978-1988	115m-bottom	±0.00 °C	f0.00 °C/a
Bothnian Sea	T	1984-1985	115m-bottom	-0.82 °C	-0.41 °C/a

SUMMARY

1. The meteorological conditions during the assessment period 1984-1988 can be characterized as variable: there have been three very cold and weak-wind winters (1984/1985 to 1986/1987) with heavy ice conditions (maximum ice coverage between 81 % and 98 %) followed by two warm and windy winters (1987/1988, 1988/1989). The winter 1989/1990 was extremely mild. The summer seasons were rather cool, the radiation poor and the winds weak at the beginning of the assessment period, but the summers of 1988 and 1989 were warm and calm.

2. The fresh water run-off into the Baltic Sea was equal to *or* higher than the mean (472 km^3/a) during the whole period with the minimum in 1986 (470 km^3/a) and the maximum in 1988 (530 km^3/a).

3. The most prominent phenomenon in the hydrography of the Baltic Sea was the continuing decrease in salinity in nearly all regions and layers. This process is mainly caused by the lack of major inflows of highly saline water during the last 13 years. The last effective major Baltic inflows occurred at the turn of 1975/1976 and in fall 1976. The small inflows in spring 1980, at the turn of 1982/1983, in spring 1986 and fall 1988 have had only effects on salinity and temperature in the deep layers of the Arkona, Bornholm and Gdaiisk Basins.

3.1 During the assessment period, a general decrease in salinity has been observed in the surface layer of all regions of the Baltic Sea starting in 1977. The mean annual trend varied from -0.02 PSU/a in major gulfs and -0.05 PSU/a in the Central Baltic Proper to -0.15 PSU/a in the Arkona and Bornholm Basins. Any salinity trends in the Kattegat and Belt Sea waters are strongly masked by the predominant annual cycle both in the surface and deep layers.

3.2 In the deep layers of the Baltic Sea the salinity decrease is more considerable since 1977. The mean annual trends during 1984-1988 are -0.05 to -0.09 PSU/a for the Gulfs of Bothnia, Finland and Riga; -0.1 to -0.2 PSU/a for the Eastern and Western Gotland Basins and the Landsort Deep (Northern Baltic Proper). The trend is as high as -0.4 PSU/a in the near-bottom layer of the Bornholm Basin.

4. Temperature and density in the deeper layers of the eastern Gotland Basin have also been decreasing considerably and the halocline and isohaline depths have been noticeably descending. Depending on diminution of vertical density gradients the vertical mixing processes between the different layers have been enlarged.

5. The current stagnation period, at least in the eastern Gotland Basin, must be regarded as one of the largest and most serious stagnation intervals ever recorded during this century and this has caused such extreme changes in the deep layers that never have been observed since the start of oceanographical observations in the Baltic Sea.

6. The transparency according to Secchi depths readings in the Northern Baltic Proper (1969-1985) show pronounced decrease compared with that during the first half of the century (1914-1939).

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Baltic Sea Environment Proceedings 35B (1990)
Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1984-1988; Background Document

2. OXYGEN, HYDROGEN SULPHIDE, ALKALINITY AND pH

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ABSTRACT

Long-term variations of oxygen conditions, specific alkalinity (A/S) and pH in the Baltic Sea waters are considered based on the BMP data and national data collected since the 1960s.

General depletion of oxygen has been observed in the near-bottom layers over the whole Baltic Sea. However, the rate of depletion and the main factors creating anoxic conditions differ regionally. Mean trend coefficients are given for different areas and subsurface layers. In the Transition area and the Arkona Basin the significant decrease in oxygen concentrations have occurred during the last 5-10 years due to increased production of organic matter followed by increased demand of oxygen for its decomposition. In the Baltic Proper the main reason for the vast oxygen deficient areas and extremely high concentrations of hydrogen sulphide, is the exceptionally long stagnation period. Due to the sinking of the halocline and advective processes, an increasing supply of oxygen into the intermediate water layers was observed in some central and northern basins. In the Bothnian Sea the lack of vertical convection down to the bottom during the winters resulted in a slight depletion of oxygen. In the Gulf of Finland no trend can be observed.

Weak negative trends in the specific alkalinity have been found in the surface waters of the Bornholm Basin, the Eastern Gotland Basin, the Northern Baltic Proper and the Åland Sea. More evident symptoms of a shift in the carbonate system due to the eutrophication and growing carbon dioxide uptake are demonstrated in pH values, increasing significantly in the euphotic zone of most of the investigated areas. The deep waters showed a constant alkalinity/salinity ratio together with marked positive trends in the long-term pH variations. Among the reasons for this phenomenon the denitrification process and weathering of rock carbonate due to acid rains are mentioned.

2.1 INTRODUCTION

S. Fonselius[4] and A. Trzosińska[1]

Oxygen, physically dissolved in the water, is of utmost importance for the life in the sea. It originates mainly from the atmosphere. The oxygen dissolves rapidly in water and the surface water is normally saturated with oxygen. The oxygen of the air is generally in equilibrium with oxygen dissolved in the uppermost water layers. Rapid temperature changes may cause a small under- or oversaturation of the surface water, because the solubility of oxygen is strongly temperature dependent. The oxygen solubility also depends on the salinity of the water, increasing slightly with decreasing salinity.

The most important factor influencing the oxygen saturation in the surface water is the primary phytoplankton production during the vegetative period. The phytoplankton removes carbon dioxide from the water during photosynthesis and produces oxygen gas. This causes an oversaturation of oxygen, which during the spring phytoplankton bloom may reach up to 120-140 %. Figure 1 shows the seasonal variations of dissolved oxygen in the surface water at the Gotland Deep, BY 15 (BMP J1), as monthly means from 1964-1987 (Fonselius 1988). The maximum of the plankton bloom which is very seldom covered in the measurements, is very short, only few days and therefore, the few measurements with extremely high saturation values may mask the trend. For these reasons oxygen trends in the surface water have not been included in the assessment work. One should, as a general rule, assume oxygen saturation or a small oversaturation as an annual mean in the surface water.

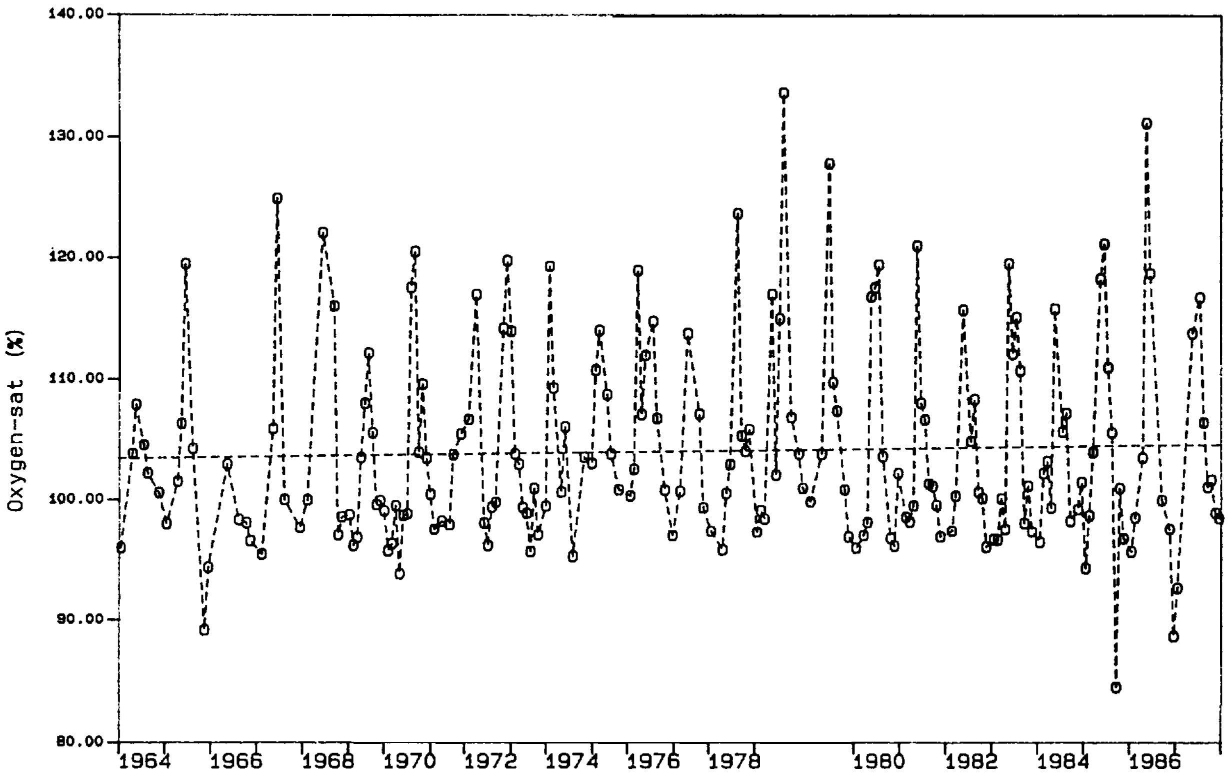


Figure 1. Monthly means of oxygen saturation values in the surface water (0-10 m) of the Gotland Deep, BY 15 (BMP Jl), from 1964 to 1987.

In the water below the surface, oxygen is consumed by bacterial oxidation of dead organic matter. The oxidation may be so rapid that the supply of oxygen through dissolution from the atmosphere cannot compensate for it. Therefore, we normally find undersaturation of oxygen in the deeper layers. In the Baltic Proper the deep water is separated from the surface layer through a permanent halocline. Above the halocline, the thermohaline convection during the winter renews the water down to the halocline. Therefore, we find seasonal variations of the oxygen saturation in these layers. Below the halocline the water is effectively isolated from exchange with the surface and the only important oxygen supply occurs through lateral water renewal. Therefore, the deep water below the halocline is always undersaturated with oxygen. If the water in the deepest parts becomes stagnant for longer periods, it may completely loose all its oxygen, especially if the phytoplankton production in the surface layers of the Baltic Sea is high. When all oxygen has been exhausted, hydrogen sulphide begins to be formed in the sediment/water interface through bacterial reduction of sulphate ions. The hydrogen sulphide is chemically dissolved in the water mainly as HS^- ions and spreads in the anoxic water layers. In presence of oxygen it will again react to form sulphate. In this reversible reaction one sulphide ion corresponds to two oxygen molecules, which gives the basis for the concept *'negative oxygen" (Fonselius 1969). Thus "negative oxygen" is the amount of oxygen equal to the amount of hydrogen sulphide produced through reduction of sulphate ions. The sulphate ion contains 4 atoms of oxygen which are used for the bacterial oxidation of organic matter and 1 atom of sulphur which is reduced from S^{6+} to S^{2-}. Multiplication with 2 of the H_2S value expressed in ml/l gives the "negative" O_2" in ml/l.

A box model of long-term dynamics of the mass balance (Savchuk 1986, Savchuk et al. 1989), was used to hindcast year-to-year variations of organic matter, inorganic nitrogen, phosphorus and oxygen in the Baltic Sea for the 1951-1982 interval. Numerical experiments let the authors suggest that the leading cause of eutrophication of the Baltic Sea is the increase of anthropogenic loads of organic matter and nutrients, while the natural changes cause background year-to-year variations only (see also Chapter 3. "Nutrients").

Alkalinity of sea water has been shown to be a function of chlorinity (Buch 1945). This implies that for consideration of any alkalinity changes caused by factors other than varying salinity, the normalized values of "specific alkalinity", expressed as A/Cl or A/S ratios, should be used instead of alkalinity itself. Strong correlations found in sea water between alkalinity, chlorinity and salinity allows also for calculation of alkalinity from the latter determinands with an accuracy which is satisfying for most oceanographic purposes. Therefore, alkalinity measurements are now seldom carried out on a routine basis. In the Oceanographical Laboratory of the Swedish Meteorological and Hydrological Institute (SMHI), however, the alkalinity of Baltic waters has been measured by the same method since the 1960s. These long data series from the international deep stations have mainly been used in the present assessment.

In the Baltic Sea the alkalinity is strongly influenced by the river water discharges, which contain relatively large amounts of carbonates. The turnover time of the Baltic Sea is approximately 30-35 years, limiting the contribution of oceanic water as an average to around 20 % in the upper layers and to 33 % in the deeper layers. Riverborne carbonates increase considerably the A/S ratio in the Baltic Sea, as compared with the almost constant value of 0.068 for the oceans. This increase is much stronger in the coastal zone than in the open sea and stronger in the upper isohaline layer than in the deep water. Table 1 shows the mean values of A/S in the surface water and bottom water of the Gotland Deep, BY 15 (BMP Jl), measured by different authors between the 1930s and 1980s.

Similar mean values have been found during 1959-1961 for the Southern Baltic Proper, viz. 0.203, for the surface waters of the Bornholm Deep and the Gdańsk Deep, 0.194, for the surface waters of the Arkona Basin (Trzosińska 1967). Respective values for the near bottom waters were 0.126 (88 m), 0.138 (106 m) and 0.108 (47 m). The A/S ratios found by Mlodziiiska (1974) in the Bay of Gdańsk, affected by the Vistula river, were 4-5 times higher. For example, the mean A/S ratio calculated for the coastal waters with salinities less than 6 PSU, amounted to 0.940.

Table 1. Comparison of Alkalinity/Salinity (A/S) mean values in the Gotland Deep, BY 15 (BMP J1) measured by different authors from 1927 to 1989.

Author	Period	Surface water	Bottom water
Buch (1945)	1927-1935	0.217	0.144
Zarins and Ozolins (1934) and Miezis and Ozolins (1940)	1933-1938	0.218	0.142
Wittig (1940)	1938	0.214	0.138
Koroleff (1954, 1957, 1958)	1954-1956	0.199	0.128
Board of Fisheries, Sweden (ICES 1966)	1958-1961	0.203	0.142
Nehring and Rohde (1967)	1965	0.206	0.134
Kremling (1969, 1970, 1972)	1966-1970	0.219	0.143
DDR IBY (ICES 1975)	1969	0.205	0.141
Poland IBY (ICES 1975)	1969	0.206	0.146
Finland IBY (ICES 1975)	1969	0.206	0.141
Denmark IBY (ICES 1975)	1969	(0.234)	0.141
SMHI Oceanogr. Laboratory (SMHI data base 1989)	1964-1987	0.206	0.142
Mean value	1927-1987	0.2090	0.1402
Standard deviation		0.0068	0.0049

All authors with the exception of Kremling (1969) used back titration techniques (e.g. Gripenberg 1936) and therefore the precision should be almost equal. Kremling used the pH method by Anderson and Robinson (1946). It has to be pointed out that the results by Koroleff which differ considerably from the other results, consist of only 3 surface values and 4 deep values taken in the years after the salt inflow in 1951, when the salinity in the Gotland Deep was still above 13.2 PSU. Kremling's results consist of 6 surface and 7 deep values. Also the low deep values by Nehring and Rohde are due to a salt inflow with salinities around 13 PSU. These values coincide very well-with Swedish measurements one month earlier. The Danish IBY surface value of 0.234 is surprisingly high and difficult to explain.

The pH of sea water is together with the alkalinity mainly used for calculating the total carbon dioxide of the water at primary production studies. The pH increases during vegetative periods due to uptake of carbon dioxide by the phytoplankton. This effect is much larger than the temperature effect, which decreases the pH with rising temperature due to increasing hydrogen ion activity (Buch and Nynäs 1939). The salinity of the water decreases the hydrogen ion activity and therefore increases the pH. This effect is also very small. Table 2 shows mean annual variations of pH in the Southern Baltic Sea.

Table 2. Long-term variations of **pH** in the southern Baltic Sea (**Trzosińska** 1990).

Reuion, station	Period	Season	Depth (m)	Mean trend coeff. pH units/a
Bornholm Deep	1969-1986	warm	0- 20	0.011[a]
BY 5 (BMP K2)	1969-1986	cold	0- 50	-0.006
	1969-1986	year	60- 90	0.001
Eastern **Gotland** Basin	1969-1986	warm	0- 20	0.012[a]
BCS-III 10 **(BMP K1)**	1969-1986	cold	0- 60	0.000
	1969-1987	year	70- 90	0.018[b]
Gdańsk Deep	1969-1986	warm	0- 20	0.027[b]
P 1 (BMP **L1**)	1979-1986	cold	0- 70	0.001
	1979-1987	year	80-108	0.024[a]

Probability of error, Student's **t-test:** a c 10 %, b < 1 %

Figure 2 shows the seasonal variations of **pH** in the surface water of the **Gotland** Deep, BY 15 (BMP J1). The summer periods show **a** maximum, because carbon dioxide is removed from the water through the phytoplankton assimilation. This raises the **pH** of the water. The reasons for variations in the deep water should be closer investigated.

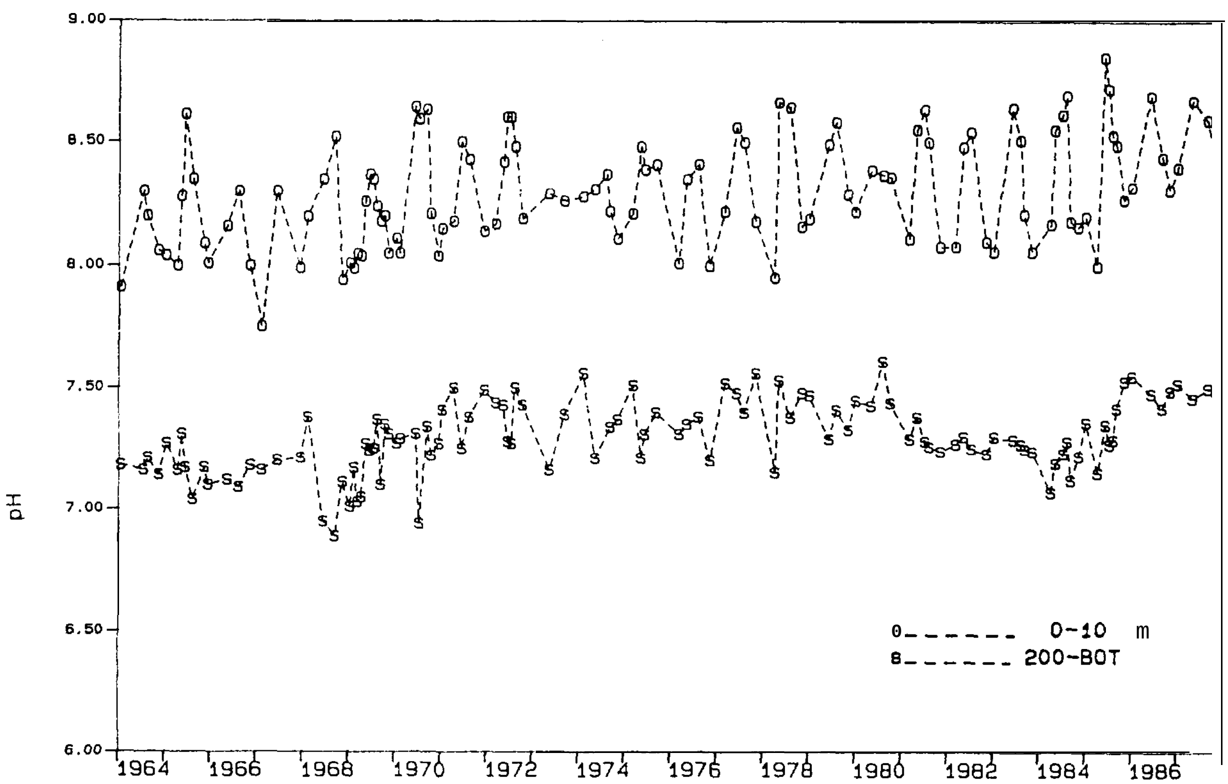


Figure 2. Monthly means of **pH** values in the surface water (0-10 m) and in the bottom water (200 m to bottom) of the **Gotland** Deep, BY 15 (BMP Jl), from 1964 to 1987.

Most of the pH measurements reported here, have been carried out by Sweden (at first by the Hydrographic Laboratory of the National Board of Fisheries, later by the Oceanographical Laboratory of SMHI). The temperature correction at the calculation of the pH was changed around 1972 (Gieskes 1969). This may have influenced the results to some extent. This has to be checked closer (Fonselius 1988).

BMP and national data have been used to assess regional conditions. Trend coefficients were calculated by means of the linear regression method. The statistical significance of the calculated results was tested according to Student's t-test. Probability levels equal to or higher than 0.95 were considered as statistically significant. Insignificant trend coefficients are shown, if necessary, in brackets.

2.2 REGIONAL ASSESSMENT OF OXYGEN CONDITIONS

2.2.1 The Kattegat, the Sound and the Belt Sea

G. Ertebjerg[8], S. Fonselius[4], H.P. Hansen', D. Körner[6] and H. Rumohr[5]

Monthly means of oxygen in ml/l from 60 m-bottom have been plotted from the station Fladen (BMP R6) from 1965-1988 (Fig. 3). A weak negative trend can be seen in the deep water for this period (Table 3). For the last five years, however, the negative trend is stronger.

Table 3. Long-term trends of monthly mean values of oxygen and negative oxygen concentrations in the Baltic Sea.

Area or station	Period	Depth m	Mean trend coefficient ml/l per year	Overall trend ml/l
Fladen	1965-1988	60-bottom	(-0.02)	(-0.48)
Landskrona Deep	1965-1988	45-bottom	(-0.003)	(-0.06)
Arkona Deep	1965-1988	45-bottom	(-0.04)	(-0.96)
Bornholm Deep	1965-1988	80-bottom	-0.06	-1.52
BY 8, By 9	1965-1988	90-bottom	(-0.007)	(-0.16)
Gotland Deep	1965-1988	100	+0.04	+0.96
Gotland Deep	1965-1988	200-bottom	-0.13	-3.04
Gdaiisk Deep	1960-1988	80	+0.055	+1.60
Gdaiisk Deep	1960-1988	101-108	-0.052	-1.51
By 27,28,29	1965-1988	125-bottom	(-0.003)	(-0.06)
Landsort Deep	1965-1988	200-300	+0.01	+0.24
Landsort Deep	1965-1988	400-bottom	(0.00)	(0.00)
BY 34,35,36	1965-1988	90-bottom	+0.05	+1.12
Åland Sea	1965-1988	200-bottom	(+0.01)	(+0.24)
Bothnian Sea	1965-1988	100-bottom	-0.03	-0.72
Bothnian Bay	1965-1988	70-bottom	(-0.01)	(-0.24)

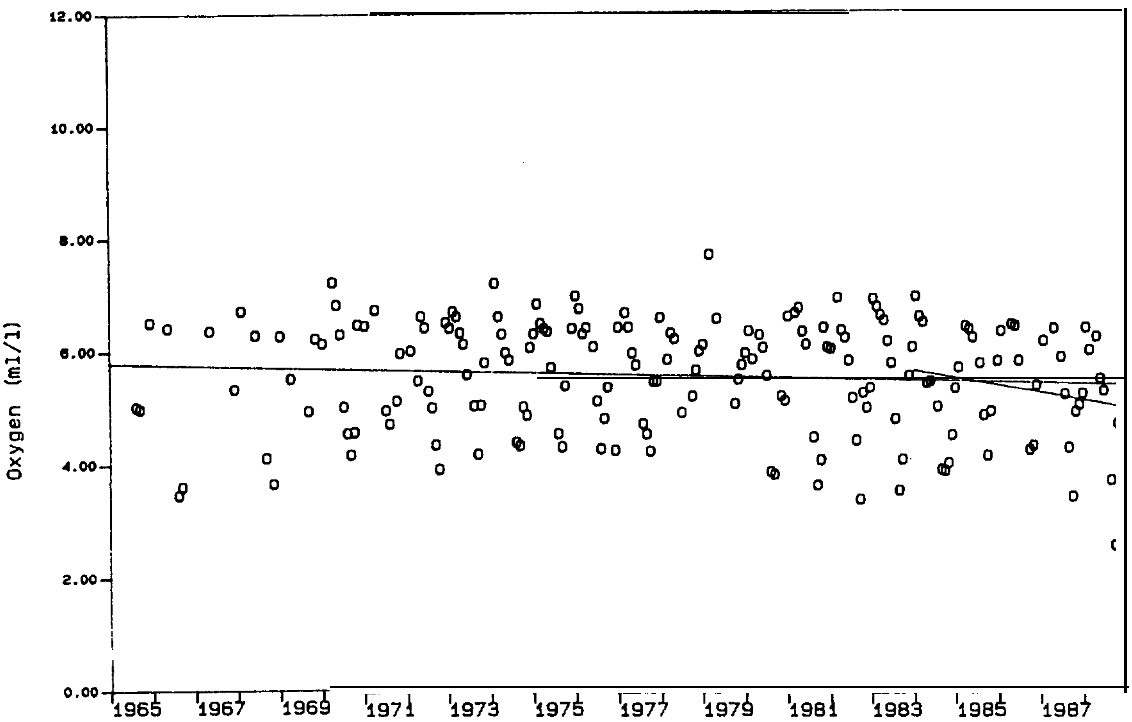


Figure 3. Long-term trends of monthly means of oxygen values (ml/l) in the bottom water (60 m to bottom) of the station Fladen (BMP R6) in the Kattegat from 1965 to 1988.

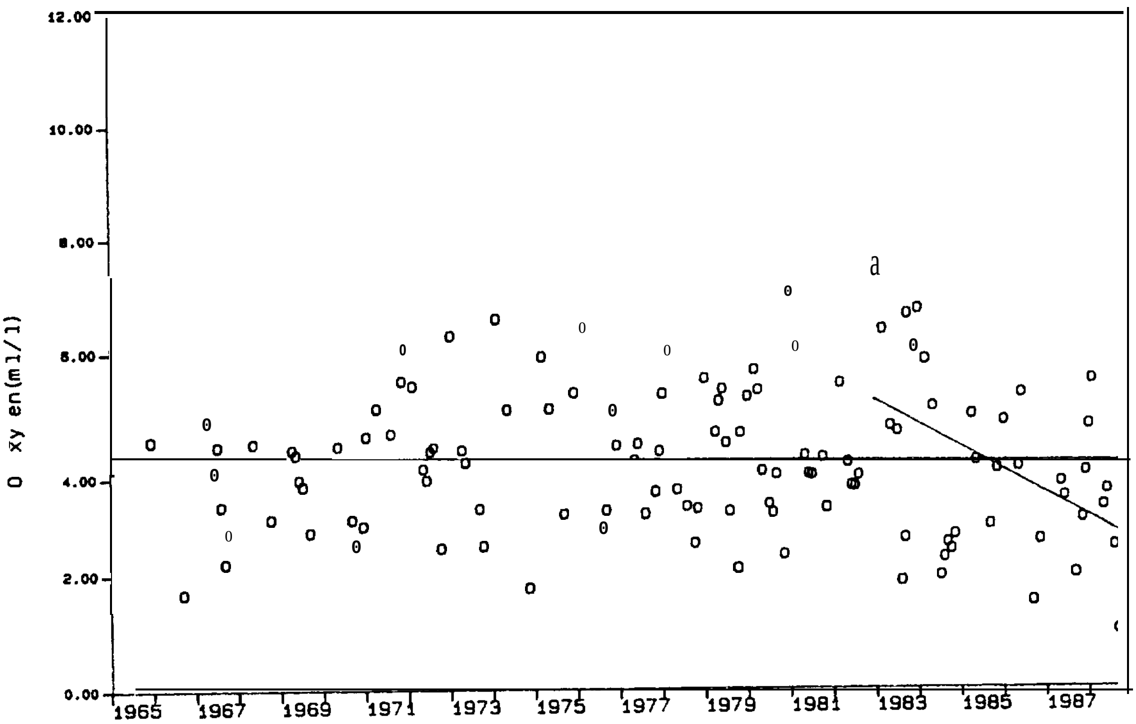


Figure 4. Long-term trends of monthly means of oxygen values (ml/l) in the bottom water (45 m to bottom) of the Landskrona Deep, station 431 (BMP Q2) in the Sound from 1965 to 1988.

In the Sound no clear trend for the whole period can be detected (Fig. 4). The figure shows monthly means of oxygen in ml/l at the Landskrona Deep (BMP Q2) from 45 m-bottom from 1965-1988 (Table 3). Also here a negative trend can be seen for the last five years.

Low oxygen concentrations have often been observed in the 1980s in the southern Kattegat, the Sound, the Belts and the Belt Sea, especially in the years 1981, 1983, 1985, 1986 and 1988. However, total oxygen-free water or hydrogen sulphide in the water has not been observed in the Kattegat, the Sound or the Great Belt. Sulphur bacteria on the sediment surface have been observed in a few minor areas of the Kattegat in 1988, and hydrogen sulphide was found in the water below the pycnocline in the Kiel Bay, the Fehmarnbelt and the Bay of Mecklenburg in September 1981, and in the Kiel Fjord in October 1986.

In the southern Kattegat the hitherto most serious oxygen deficiency was observed in the autumn 1988 with oxygen concentrations below 3 ml/l for more than 2 months (Fig. 5), and often below 1 ml/l. The effect was that fish caught in nets and many bottom invertebrates died. Among these the population of *Nephrops norvegicus* was seriously harmed.

A statistical model analysis of all oxygen data from the period 1974-1987 and the depths 20 m-bottom from the stations Halsskov Rev in the Great Belt, the Landskrona Deep in the Sound and Gniben and Anholt East in the Kattegat shows equal seasonal variations and development with time. Therefore, all the observations were pooled. The analysis shows in general (Fig. 6) that the autumn oxygen minimum has gradually become lower, and that another oxygen minimum in the spring, after the spring phytoplankton bloom, has gradually developed during the period 1974-1987.

Although the oxygen concentrations during the spring oxygen minimum generally are much higher than during the autumn oxygen minimum, the spring minimum has caused the death of fish caught in nets along the northern coast of Zealand in 1987 and 1988.

Total lack of oxygen has been observed in the bottom waters of some parts of the Kiel Bay occasionally during the last 100 years. However, zero oxygen concentration in two consequent years was reported only once before World War II and again for 1969 to 1971. The last decade (1979 to 1989) is characterized by a significant negative trend in the oxygen concentrations in the bottom water during the autumn minimum (August-September; 25 m to bottom, trend coefficient -0.12 ml/l per year). The negative trend in oxygen concentrations (Fig. 7) is confirmed by a corresponding negative trend in pH (see Fig. 29). Observations of hydrogen sulphide formation have been reported more frequently for most of the Kiel Bay stations during the last decade.

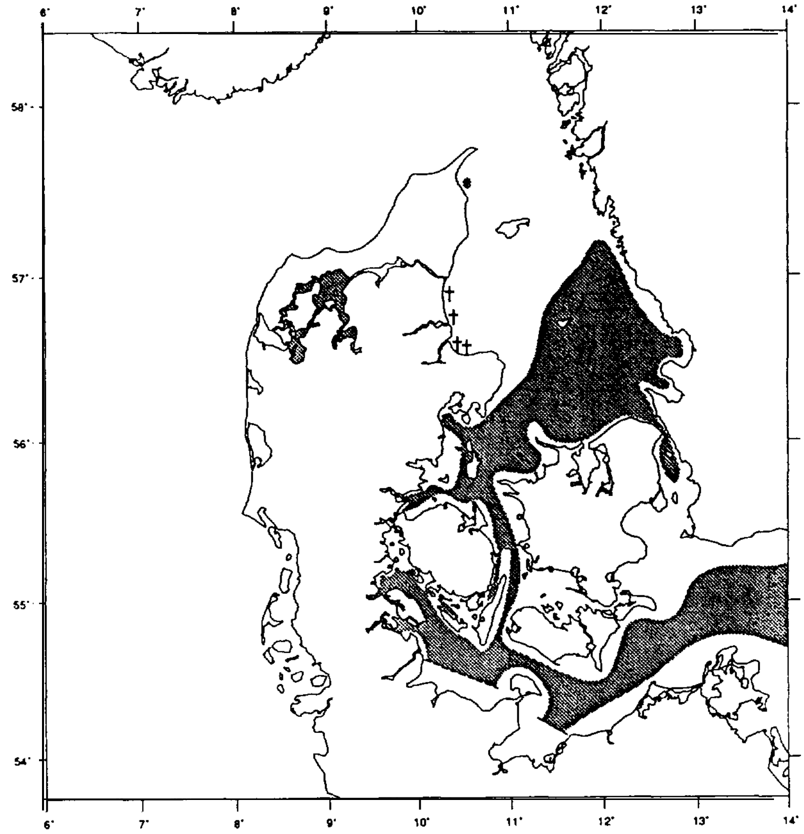


Figure 5. Map of Kattegat, Belt Sea and Arkona Sea showing areas with oxygen depletion in the fall 1988. Shaded areas had oxygen concentrations below 3 ml/l. Crosses show places with dead bottom animals washed ashore 12-13 October 1988.

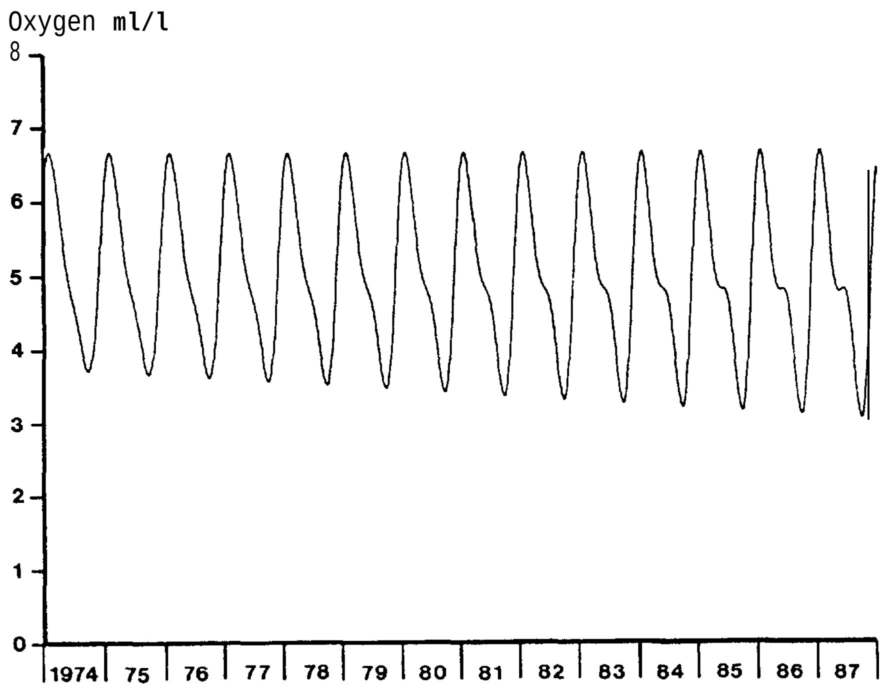


Figure 6. Modelling of the development of the oxygen conditions (ml/l) in the bottom water of the northern Sound, the southern Kattegat and the Great Belt during the period 1974-1987.

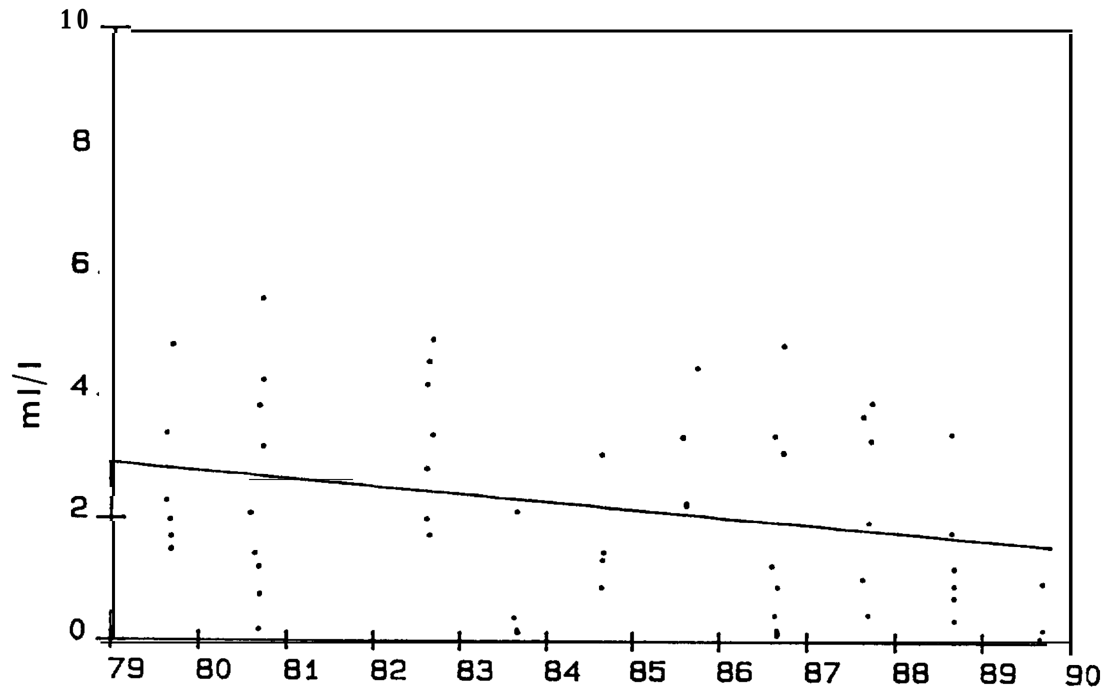


Figure 7. Long-term trends of oxygen concentrations (ml/l) in the bottom water (25 m to bottom) of the Kiel Bay during the oxygen autumn minimum (August-September) from 1979 to 1990.

2.2.2 The Baltic Proper

Arkona Basin
S. Fonselius[4]

In the deep water the variations of the oxygen concentrations are large due to the fact that the halocline is located very close to the bottom and that therefore many of the samples have been taken above the halocline; considerable seasonal variations also influence the results. A decrease in oxygen concentrations, however, seems to occur in the deep water of the Arkona Basin, BY 2 (BMP K4), from 1965-1988 (Fig. 8), but the **overall trend** (Table 3) as well as the sub-trend for 1977-1988 (Table 4) are not significant. For the last 5 years the negative trend is clear.

Table 4. Characteristic changes in oxygen conditions (Matthäus 1990).

Area or station	Period years	Depth m	Mean trend coeff. ml/l per year	Overall trend ml/l
Arkona Basin	1977-1988	45	not signific.	not signific.
Bornholm Basin	1977-1988	80	-0.149	-1.8
Gotland Deep BY 15	1977- Apr.1982	100	-0.306	-1.6
" " "	Nov.1983-1988	100	-0.406	-2.0
" " "	1977-1988	200	-0.404	-4.7

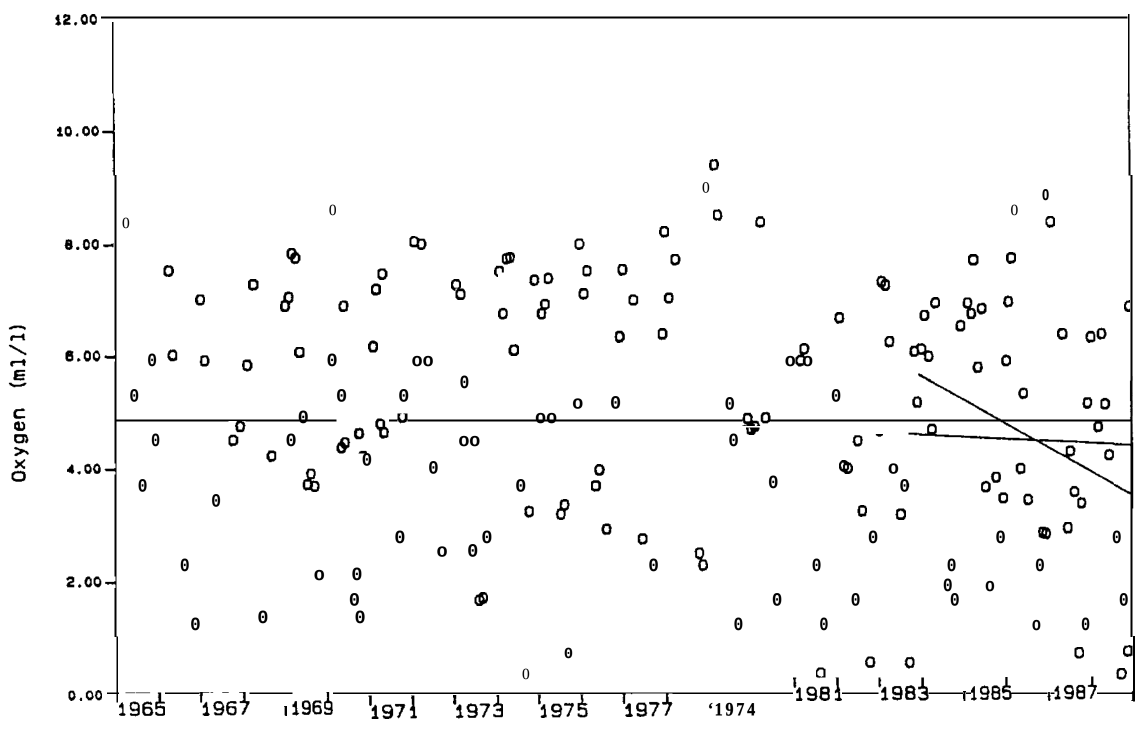


Figure 8. Long-term trends of monthly means of oxygen values (ml/l) in the bottom water (45 m to bottom) of the Arkona Deep, BY 2 (BMP K4) from 1965 to 1988.

Bornholm Basin

S. Fonselius and D. Nehring[7]

In the Bornholm Basin clear inflows of oxygen rich water can be seen with intervals of about 2-4 years. Hydrogen sulphide is occasionally formed in the bottom water (Fonselius 1984). The graph shows monthly mean values of oxygen and hydrogen sulphide (negative oxygen) at 80 m-bottom from 1965-1988 as means for the stations BY 4 and BY 5 (BMP K2). The occurrence of hydrogen sulphide has become more frequent during recent years. A clear negative trend can be seen for the period (Fig. 9), which amounts to -1.8 ml/l between 1977 and 1988 (cf. Table 4). The more negative trend for the last 5 years exists also here (see also Table 3).

Eastern Gotland Basin

S. Fonselius' and W. Matthäus[7]

Figure 10 shows the conditions 1965-1988 in the southern part of the basin at the stations BY 8 and BY 9 from 90 m to the bottom. No significant trend can be found (Table 3). For shorter periods there are clear trends indicating inflows of new water. For the five last years there is a very clear negative trend. Figure 11 shows the development in the Gotland Deep, BY 15 (BMP J1), during the same period from 200 m to the bottom (see also Tables 3 and 4). Also here several inflows of new water can be found. The trend shows a significant decrease in oxygen content during the whole period and almost stable anoxic conditions during the last 5 years (1984-1988). The Gotland Deep is located in a semi-stagnant basin, where the bottom water is irregularly renewed through horizontal exchange along the bottom. The water renewals can be seen as maxima in the oxygen concentration in the figure. We can see that the last major inflow which was significant for the Gotland Deep water renewal, occurred at the beginning of 1977 and that since then the bottom water slowly lost its oxygen, forming hydrogen sulphide (Matthaus 1986, 1987, Fonselius 1988). One of the reasons for the long-stagnation period after 1977 seems to be the inflow in January 1977 described by Fonselius (1977), which raised the temperature to more than 7°C and the salinity to more than 13 PSU below 200 m. The saline water inflows at the turn of 1982-1983 and in the spring of 1986 (see Chapter 1 "Hydrography") affected the intermediate layers only. This can be seen in Figure 12 where the oxygen trend below the halocline at around 100 m is shown. The trend is clearly positive (Table 3) and the above-mentioned inflows can be seen as maxima in the oxygen concentration. The reason for the positive trend can be the increased advection of oxygen-rich water in intermediate depths. It seems, however, to be more probable that the vertical exchange across the primary halocline has increased due to the decreased stability of the stratification and the significant shifting of the halocline centre to greater depths (Matthaus 1990). For the separate periods of the current stagnation negative sub-trends could be found in the intermediate waters (Table 4; Fig. 12).

The stagnation period in the Eastern Gotland Basin started with the highest temperatures ever recorded in the bottom water (> 7°C) and led to the formation of the highest concentrations of hydrogen sulphide (> 3 mg/l) ever measured at the 200 m level of the Gotland Deep, BY 15 (BMP J1). Compared to earlier extreme periods (cf. Matthaus 1986, 1987) the current stagnation period caused the greatest decreases in oxygen concentrations which ever have been observed since the beginning of oceanological observations in the Baltic Sea (Matthaus 1990).

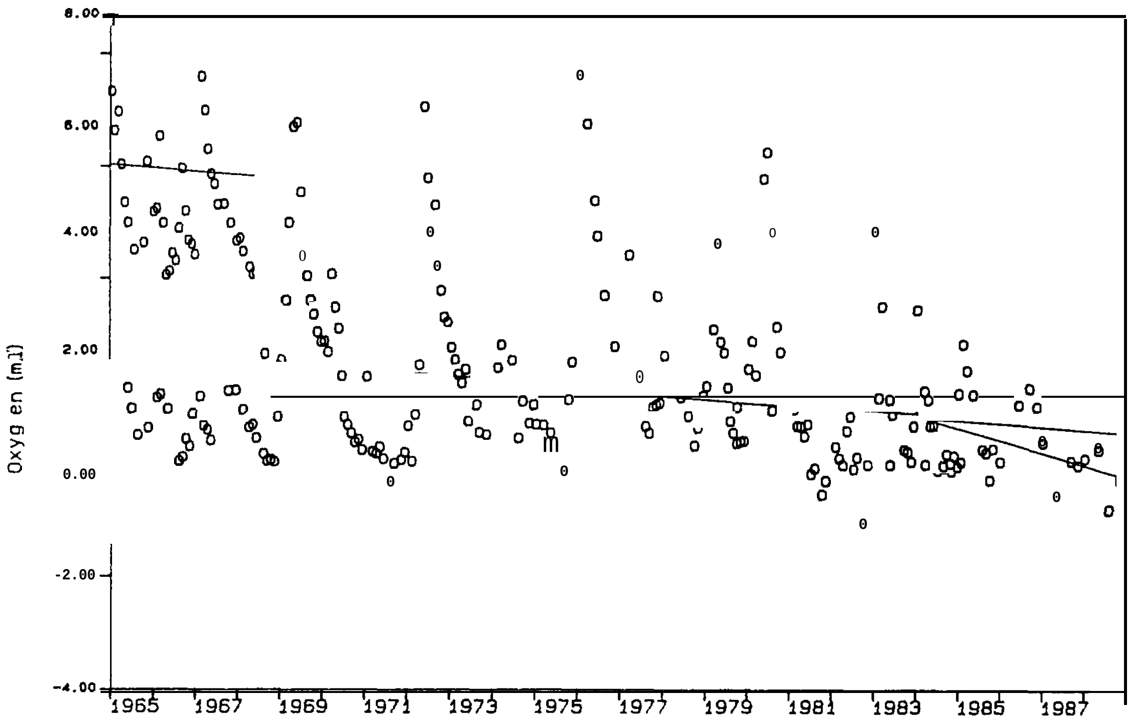


Figure 9. Long-term trends of monthly means of oxygen values (ml/l) in the bottom water (80 m to bottom) of the Bornholm Basin from 1965 to 1988. Mean values of the stations BY 4 and BY 5 (BMP K2) have been used.

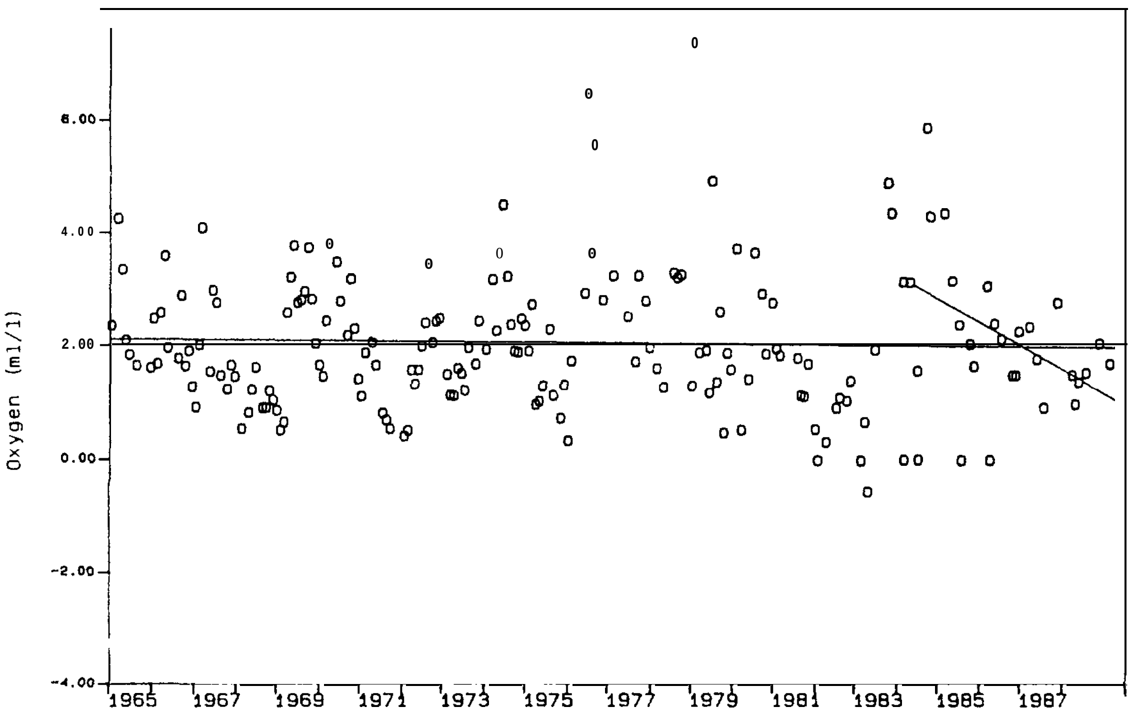


Figure 10. Long-term trends of monthly means of oxygen values (ml/l) in the bottom water (90 m to bottom) in the southern part of the Eastern **Gotland** Basin from 1965 to 1988. Mean values of stations BY 8 and BY 9 have been used.

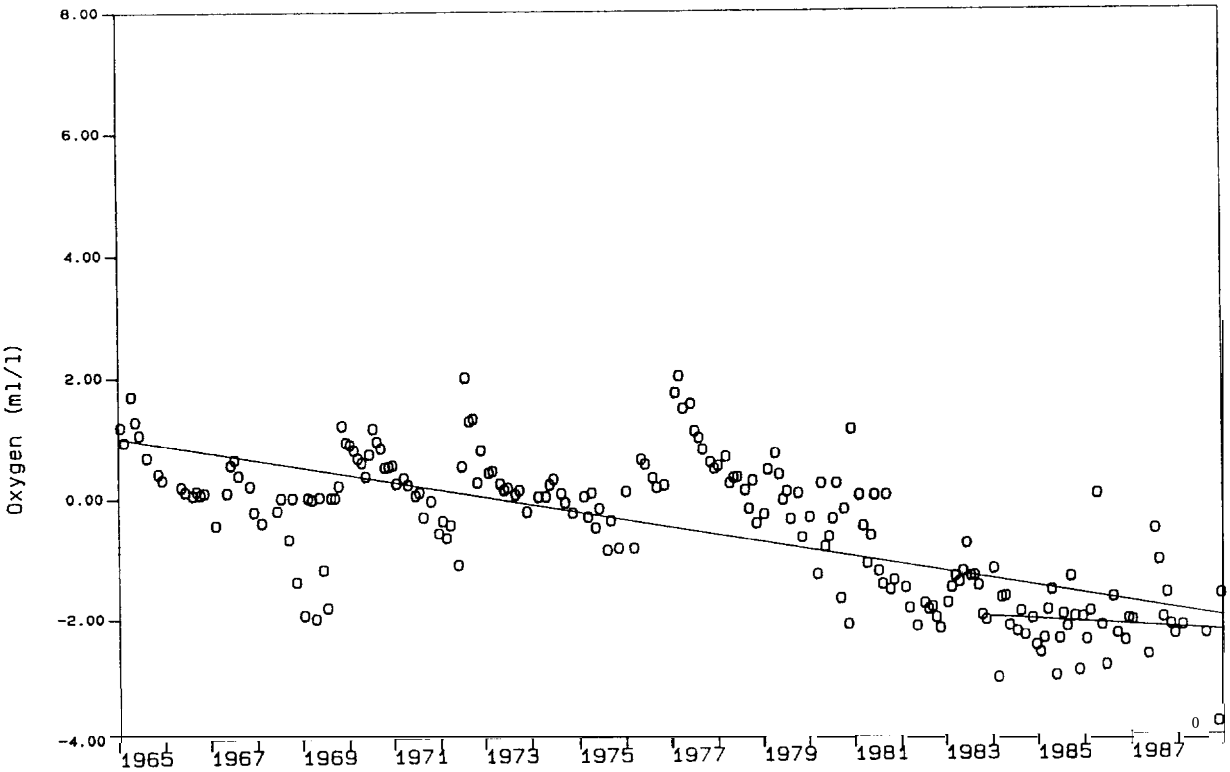


Figure 11. **Long-term trends of monthly means of oxygen and negative oxygen values** (ml/l) in the bottom water (200 m to bottom) of the Gotland Deep, BY 15 (BMP Jl), from 1965 to 1988.

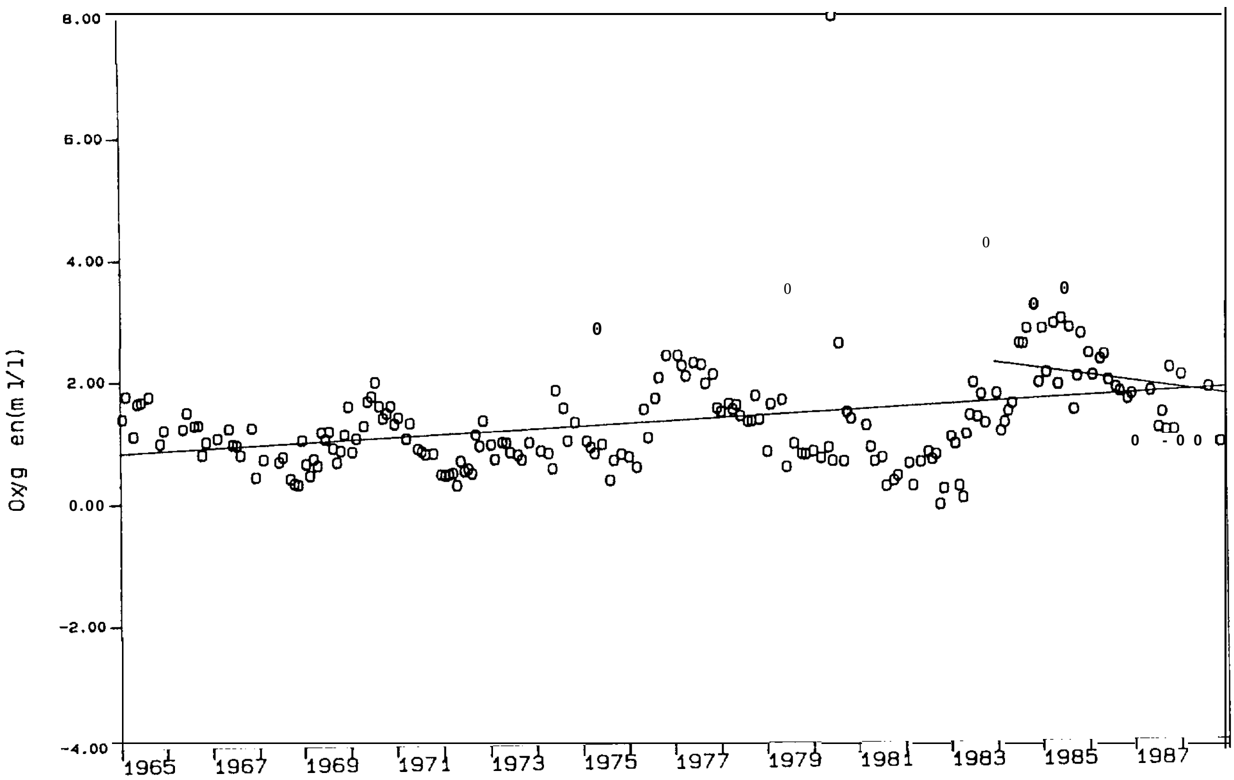


Figure 12. Long-term trends of monthly means of oxygen values (ml/l) in the intermediate water (95-100 m) of the Gotland Deep, BY 15 (BMP Jl), from 1965 to 1988.

Gdansk Basin
B. Cyberska[1]

The Gdańsk Basin is a sub-basin of the Eastern Gotland Basin. In the deep layers of the Gdaiisk Deep, Station P 1 (BMP L1), the overall negative trend in the oxygen concentration during 1960-1988 is in good agreement with the results of the period 1960-1983 (Cyberska and Trzosiiiska 1984). In the near-bottom water (100-108 m) the rate of oxygen depletion was approximately 0.05 ml/l per year or 0.4 % of saturation per year during the last three decades (Tables 3 and 5). Due to the advection of oxygen-rich water after the 1975/1976 oceanic inflow, this 30 year period can be subdivided into two periods of different rates of oxygen depletion (Fig. 13 a). During the present prolonged stagnation the oxygen depletion rate has considerably accelerated (0.14 ml/l per year). Later inflows in 1977, 1982/83 and 1986 affected the bottom waters of the Gdaiisk Deep to a much smaller extent, although their influence can be traced there as peaks in the oxygen concentration values, as disappearance of hydrogen sulphide during 1983-1986, and as a significant correlation between salinity and oxygen concentrations. Hydrogen sulphide appeared there again in the last three years (1987-1989) in very high concentrations, up to 45 µmol/l (1.5 mg/l). In 1989 the hydrogen sulphide-containing water also penetrated into the central part of the Bay of Gdańsk.

Contrary to the near-bottom water, the intermediate layers of the Gdaiisk Deep showed a highly significant positive oxygen trend during the past three decades (Fig. 13 b, Tables 3 and 5), with the negative sub-trends marked during the inter-inflow periods. For explanation of this positive trend, also observed in the Gotland Deep, see paragraph on the eastern Gotland Basin.

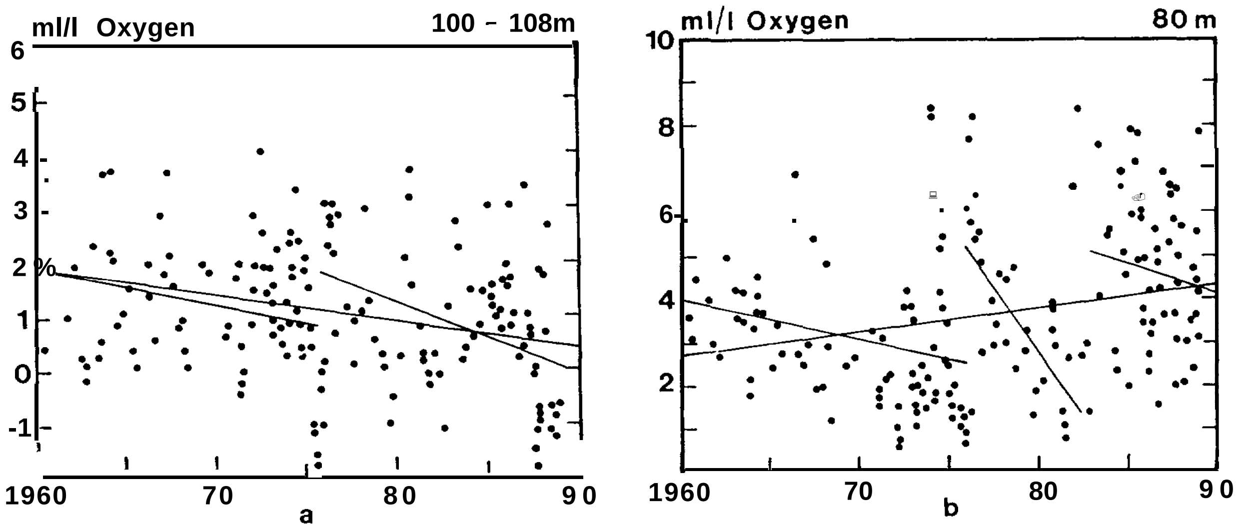


Figure 13 a. Oxygen and negative oxygen values (ml/l) in the bottom water (101-108 m) of the Gdańsk Basin at the station P 1 (BMP L1) from 1960 to 1989.

b. Oxygen values (ml/l) in the intermediate water (80 m depth) of the Gdaiisk Basin at the station P 1 (BMP L1) from 1960 to 1989.

Table 5. Characteristic changes in the oxygen conditions in the Gdańsk Deep, P 1 (BMP L1).

Period	Depth (m)	Mean trend coeff. (ml/l per year)	Overall trend (ml/l)
1960-1988	80	0.055	1.60
1960-1975	80	-0.099	-1.48
1976-1981	80	-0.649	-3.89
1982-1988	80	(-0.122)	(-0.85)
1960-1988	101-108	-0.052	-1.51
1960-1975	101-108	-0.064	-1.02
1976-1988	101-108	-0.143	-1.86

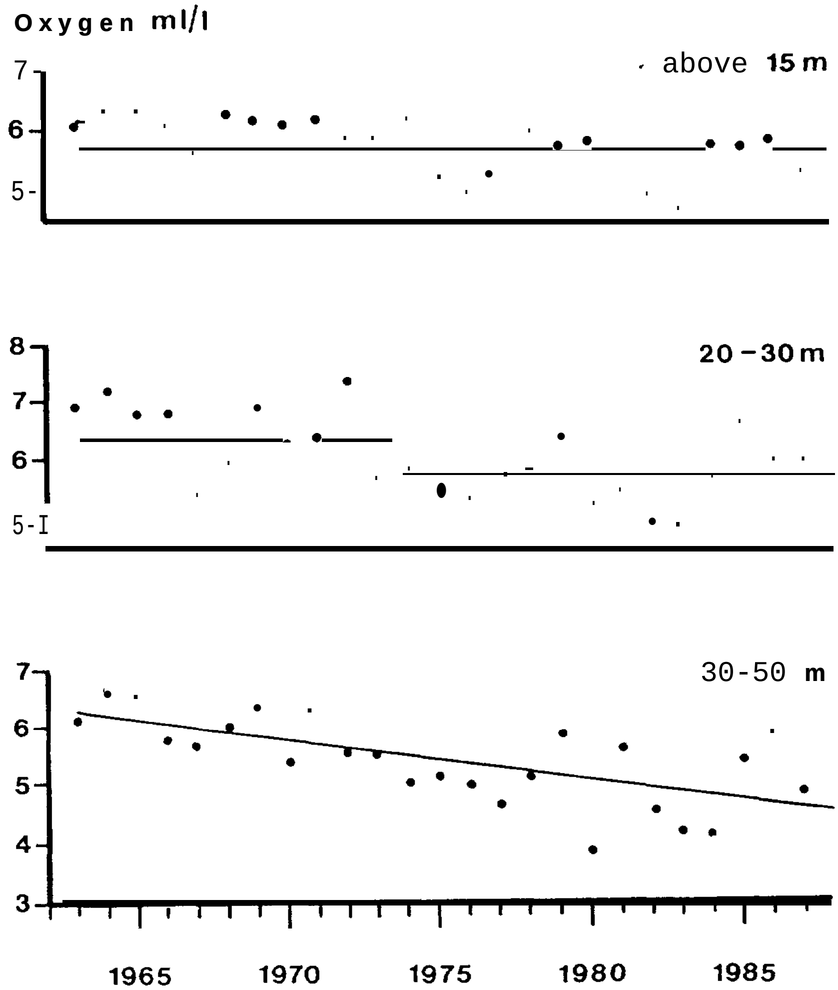


Figure 14. Long-term variations of oxygen concentrations (ml/l) in different water layers of the Gulf of Riga in August from 1963 to 1987.

Gulf of Riga
V. Berzins[3]

Long-term variations of the oxygen concentrations in the near-bottom water layer in the central part of the Gulf of Riga show significant negative trends. During the last 24 years (1963-1986) the mean decrease in summer (August) amounted to about 0.07 ml/l per year. The rate of decrease oscillated, however, with a period of 2-7 years, due to periodical changes in the density stratification, connected with the inflows of more saline Baltic water (Fig. 14).

Northern Baltic Proper
S. Fonselius[4]

The Landsort Deep, BY 31 (BMP H3), is representing the Northern Baltic Proper (Fig. 15). The figure shows the oxygen trend from 1965-1988 from 400 m to the bottom. No clear trend can be found for the period. During the last five years the conditions, however, have improved. The water inflows at intermediate depths seem to have penetrated down into the Landsort Deep (Table 3). Figure 16 shows monthly means of mean values for the stations BY 27, BY 28 (BMP H2) and BY 29 from 125 m-bottom during the same period (Table 3). No trend can be found, with exception for the last 5 years, which show a weak but significant negative trend.

Western Gotland Basin
S. Fonselius[4]

For this basin the stations BY 34, BY 35 and BY 36 have been used. Figure 17 shows mean values of these stations from 1965-1988 from 90 m to the bottom. A positive highly significant trend can be seen here, due to the previously discussed inflow of water at intermediate levels (Table 3). The density of the bottom water is here low enough for this water to penetrate down to the bottom.

2.2.3 Gulf of Finland
M. Perttilä[2]

Oxygen development in the Gulf of Finland depends partly on the development in the Baltic Proper due to lack of a sill between the Baltic Proper and the Gulf of Finland. In the water layer from 50 m down to the bottom, no significant trends can, however, yet be seen. Most of the BMP data has been collected at the station LL 7 (BMP F3), where the halocline, usually observed at single CTD casts, is very variable and thus its effects on salinity and therefore also on oxygen concentrations are obscured in the long-term scatter plot of oxygen vs. time (Fig. 18).

No consistent oxygen depletion has, however, been observed in the Gulf of Finland during the period of inspection.

2.2.4 Gulf of Bothnie
S. Fonselius[4]

No significant trend can be found at the station F 64 (BMP D1) (Table 3).

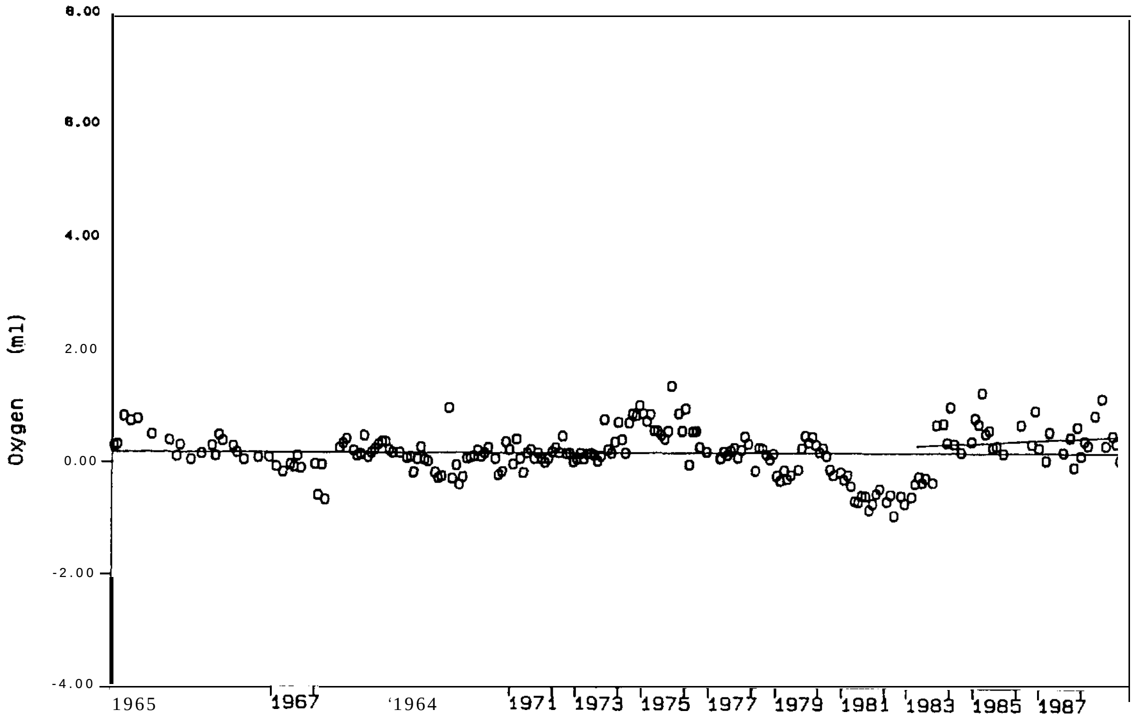


Figure 15. Long-term trends of monthly means of oxygen and negative oxygen values (ml/l) in the bottom water (400 m to bottom) of the **Landsort** Deep, BY 31 (BMP H3) from 1965 to 1988.

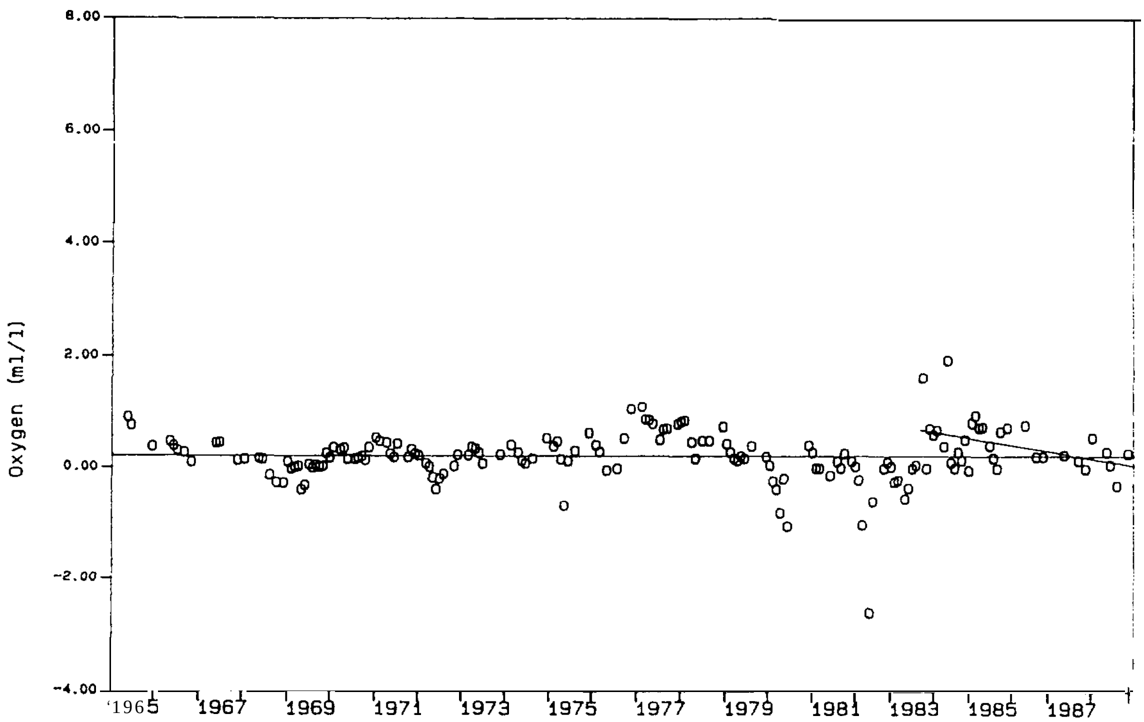


Figure 16. Long-term trends of monthly means of oxygen and negative oxygen values (ml/l) in the bottom water (125 m to bottom) of the eastern part of the Northern Baltic Proper from 1965 to 1988. Mean values of the stations BY 27, BY 28 (**BMP** H2) and BY 29 have been used.

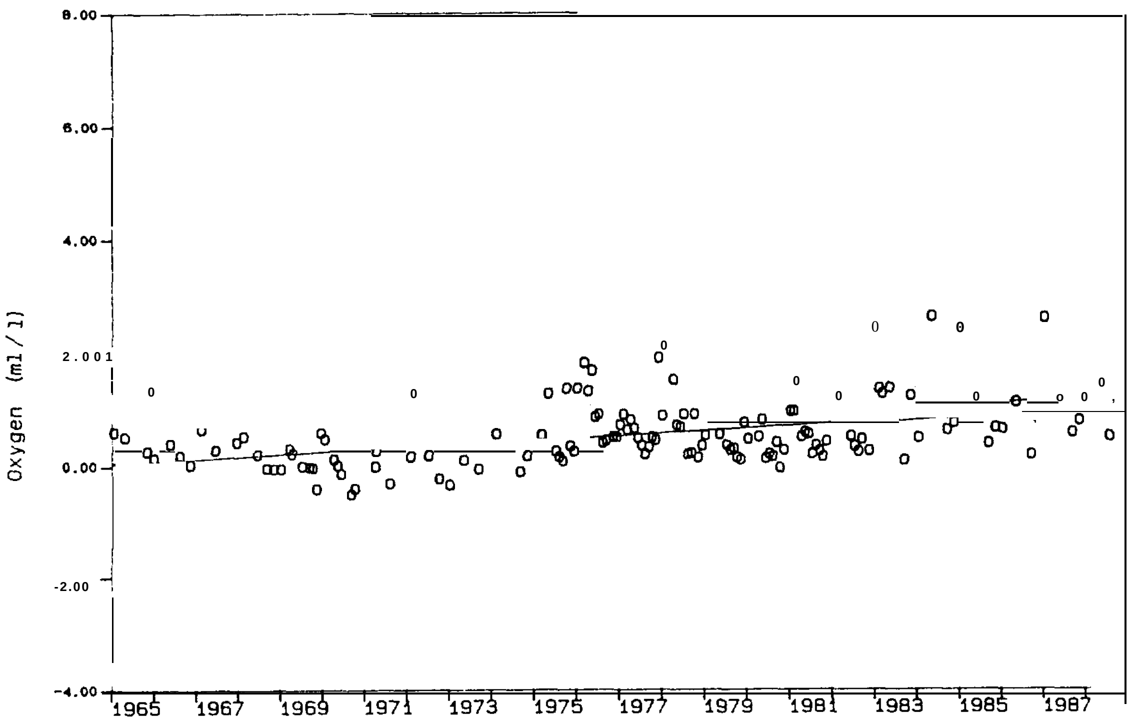


Figure 17. Long-term trends of monthly means of oxygen and negative oxygen values (ml/l) in the bottom water (90 m to bottom) of the Western Gotland Basin from 1965 to 1988. Mean values of stations BY 34, BY 35 and BY 36 have been used.

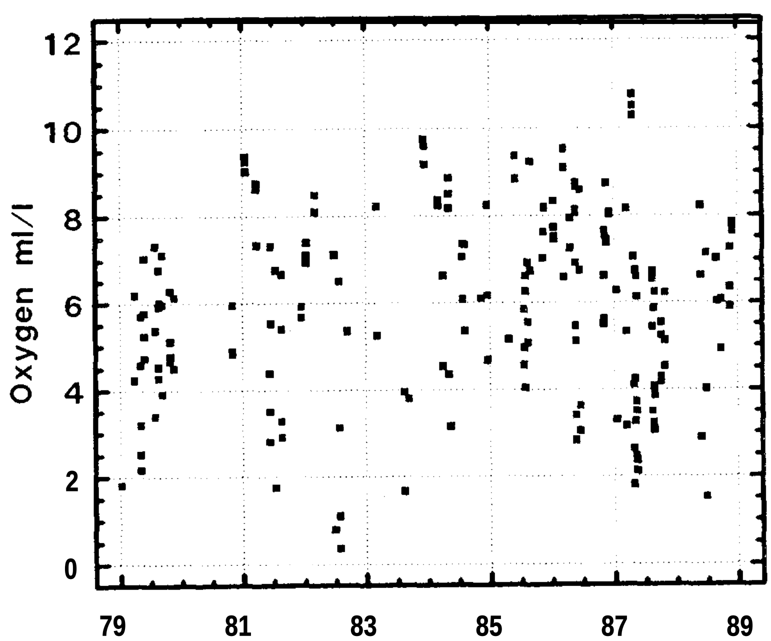


Figure 18. Oxygen values (ml/l) at station LL 7 (BMP F3) in the Gulf of Finland from 50 m to bottom during the period 1979-1989.

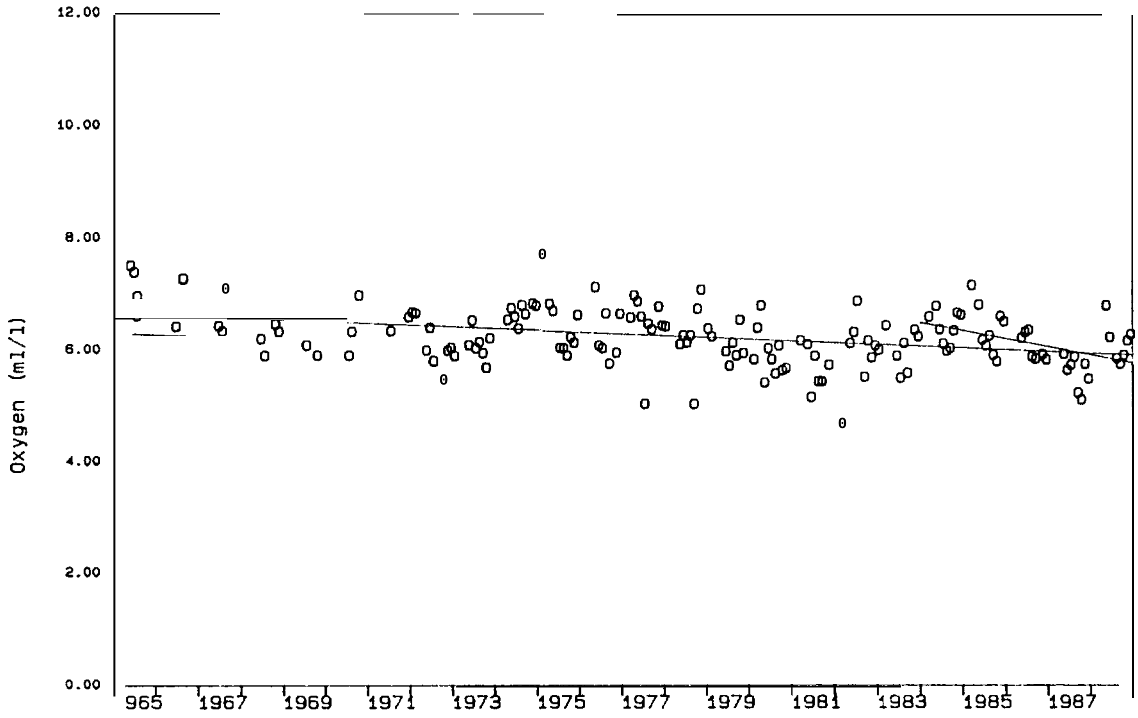


Figure 19. Long-term trends of monthly means of oxygen values (ml/l) in the bottom water (100 m to bottom) of the Bothnian Sea from 1965 to 1988. Mean values of all deep stations have been used.

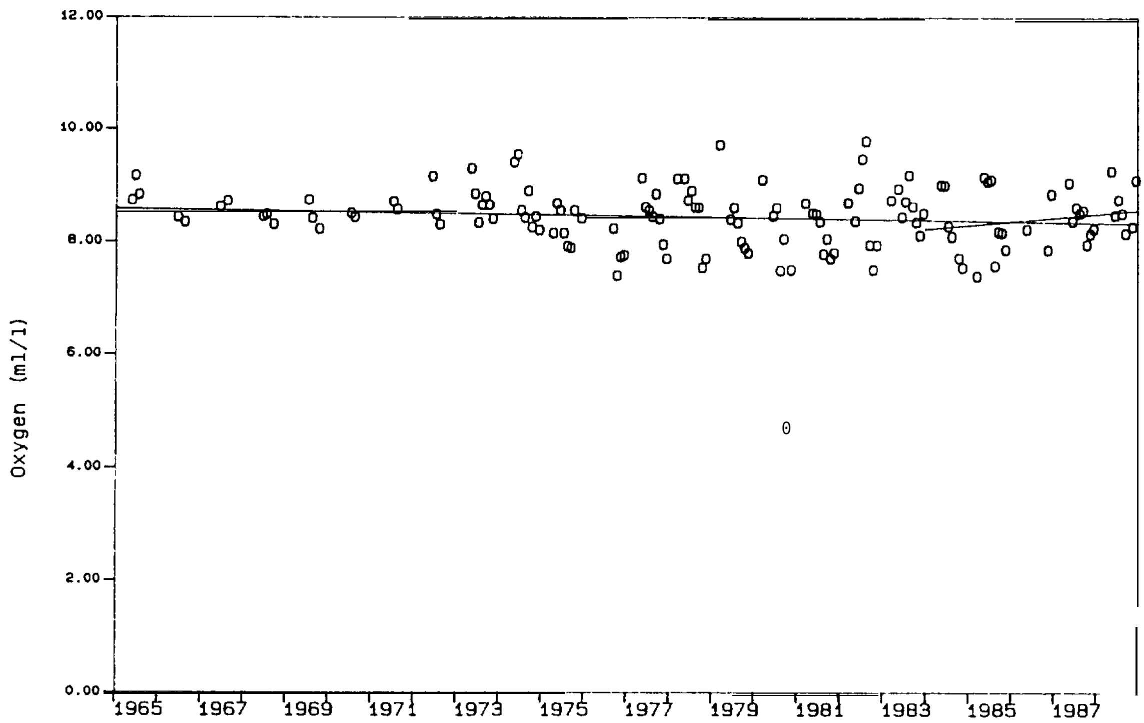


Figure 20. Long-term trends of monthly means of oxygen values (ml/l) in the bottom water (70 m to bottom) of the Bothnian Bay from 1965 to 1988. Mean values of all deep stations have been used.

Bothnian Sea

Figure 19 shows the oxygen conditions in this basin from 1965-1988 as means of several representative stations from 100 m-bottom. The trend is clearly negative during the period (Table 3). The last five years show a larger decrease of the oxygen concentrations.

Bothnian Bay

Figure 20 has also been drawn using mean values from several representative stations showing the oxygen concentration from 70 m-bottom 1965-1988. The number of observations from single stations is too small, especially during the 1960s for obtaining reliable results. No significant trend can be found (Table 3). The large supply of fresh water and the weak stratification of the water prevents the basin from developing signs of stagnation.

2.3 REGIONAL ASSESSMENT OF THE ALKALINITY

S. Fonselius[4]

For the assessment A/S relations without the constant term, have been used. Annual mean values have been calculated for all stations.

2.3.1 The Uattegat and the Sound

Annual means of A/S at the Kattegat station Fladen (BMP R6) from 1965-1988 for the surface water (O-10 m) and the deep water (50 m - bottom) have been computed. No significant trends can be found, even if there are indications of a weak negative trend in the surface water during the last 5 years (Table 6).

In the Sound (Landskrona Deep, Station 431; BMP Q2) the surface values are very scattered, but long-term negative trends can be found in the surface water and in the bottom water. The reason for the scattering in the surface water is obviously the strong influence from river discharge with a high alkalinity. The trends are not significant (Fig. 21 and Table 6). For the last 5 years (1984-1988) very weak positive insignificant trends can be detected.

2.3.2 The Baltic Proper

Arkona Basin

Negative trends could be found both in the surface water and in the deep water at the station BY 2 (BMP K4) (Table 6), but these are not significant.

Bornholm Basin

In the Bornholm Basin, mean of stations BY 4 and BY 5 (BMP K2) has been calculated. A weak negative trend could be detected at O-10 m but below 80 m no trend could be found (Table 6).

Table 6. Long-term trends of annual means of specific alkalinity (A/S) in the Baltic Sea.

Area or station	Period	Depth m	Mean trend coefficient A/S units/a	Overall trend A/S units
Fladen	1965-1988	0-10	(-0.0001)	(-0.002)
Fladen	1965-1988	60-bottom	(-0.0002)	(-0.004)
Landskrona Deep	1965-1988	0-10	(-0.0005)	(-0.012)
Landskrona Deep	1965-1988	45-bottom	(-0.0003)	(-0.008)
Arkona Deep	1965-1988	0-10	(-0.00025)	(-0.006)
Arkona Deep	1965-1988	45-bottom	(-0.00025)	(-0.006)
Bornholm Deep	1965-1988	0-10	(-0.00025)	(-0.006)
Bornholm Deep	1965-1988	80-bottom	(0.0000)	(0.000)
BY 8, BY 9	1965-1988	0-10	-0.0005	-0.012
BY 8, BY 9	1965-1988	90-bottom	(0.0000)	(0.000)
Gotland Deep	1965-1988	0-10	-0.0003	-0.008
Gotland Deep	1965-1988	200-bottom	(0.0000)	(0.000)
BY 27, 28, 29	1965-1988	0-10	(-0.00025)	(-0.006)
BY 27, 28, 29	1965-1988	125-bottom	+0.0003	+0.008
Landsort Deep	1965-1988	0-10	(-0.0002)	(-0.004)
Landsort Deep	1965-1988	400-bottom	(+0.00025)	(+0.006)
BY 34, 35, 36	1965-1988	0-10	-0.0005	-0.012
BY 34, 35, 36	1965-1988	90-bottom	(+0.0003)	(+0.008)
Åland Sea	1965-1988	0-10	(+0.0002)	(+0.004)
Åland Sea	1965-1988	200-bottom	(0.000)	(0.000)
Bothnian Sea	1965-1988	0-10	(-0.0003)	(-0.008)
Bothnian Sea	1965-1988	100-bottom	(0.0000)	(0.000)
Bothnian Bay	1965-1988	0-10	(-0.0004)	(-0.010)
Bothnian Bay	1965-1988	70-bottom	(0.0000)	(0.000)

Eastern Gotland Basin

Here results from the Gotland Deep BY 15 (BMP J1) only will be used because the results from the stations BY 8 and BY 9 are very similar (Table 6). Figure 22 shows monthly means for A/S in the surface water (0-10 m) and the deep water (200 m - bottom) from 1965-1988. Only in the surface water a very weak but significant negative trend can be seen. For the last 5 years both levels show a positive trend.

Northern Baltic Proper

The Landsort Deep, BY 31 (BMP H3), is representative for the Northern Baltic Proper. Figure 23 shows the annual means of A/S at 0-10 m and from 200 m - bottom at the Landsort Deep. Also here the trend is negative in the surface water, but we may see a weak positive trend in the deep water. The mean of the stations BY 27, BY 28 (BMP H2) and BY 29 show a similar result (Table 6). The only significant trend was found in the deep water of BY 27, 28 and 29.

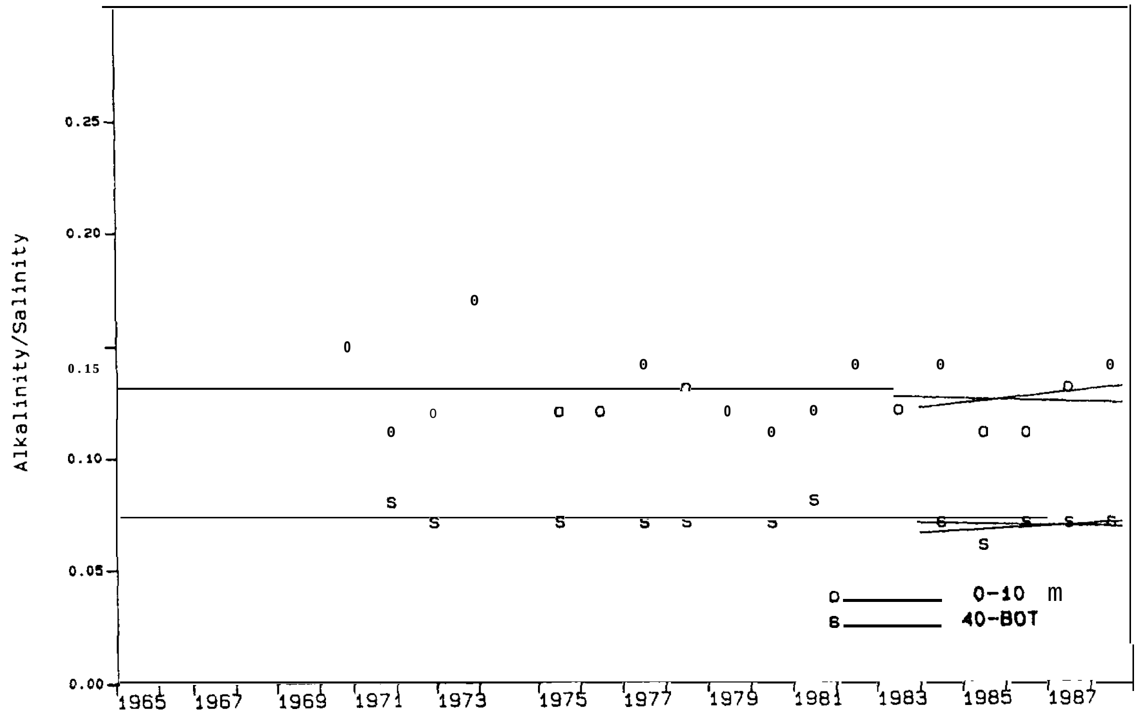


Figure **21.** Long-term trends of annual means of A/S in the surface water (0-10 m) and the bottom water (40 m to bottom) of the Landskrona Deep (BMP Q2) from 1970 to 1988.

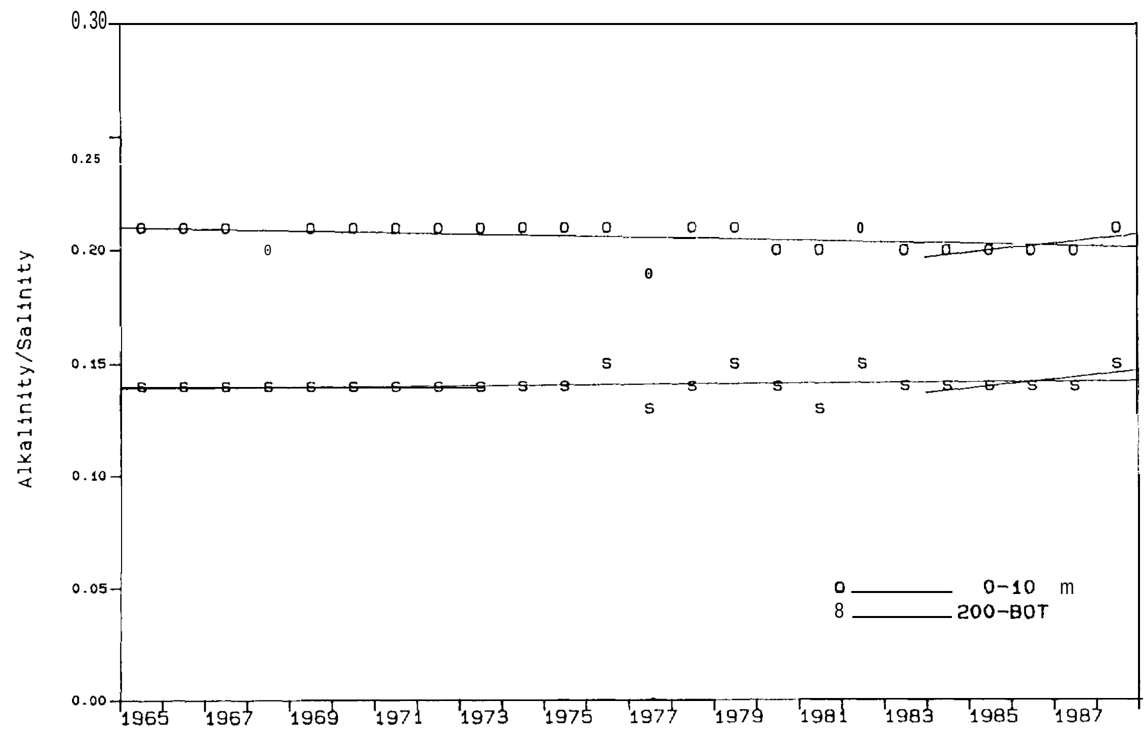


Figure 22. Long-term trends of annual means of A/S in the surface water (0-10 m) and the bottom water (200 m to bottom) of the Gotland Deep, BY 15 (BMP J1) from 1965 to 1988.

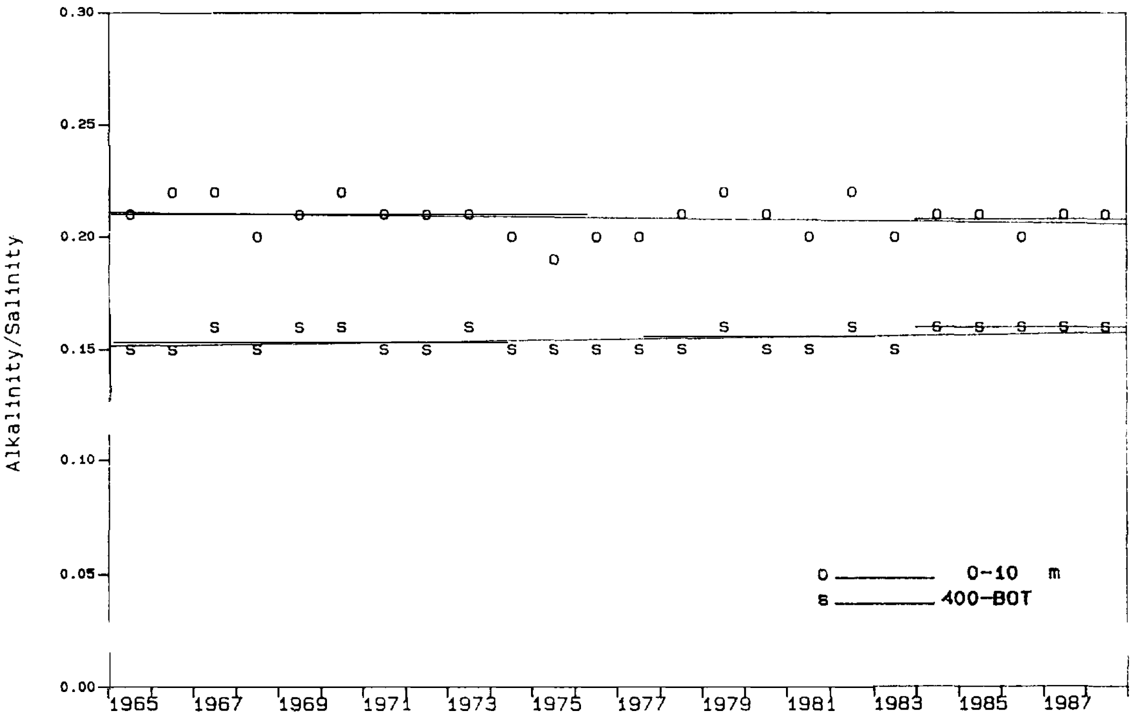


Figure 23. Long-term trends of annual means of A/S in the surface water (0-10 m) and the bottom water (400 m to bottom) of the **Landsort** Deep, BY 31 (BMP H3), from 1965 to 1988.

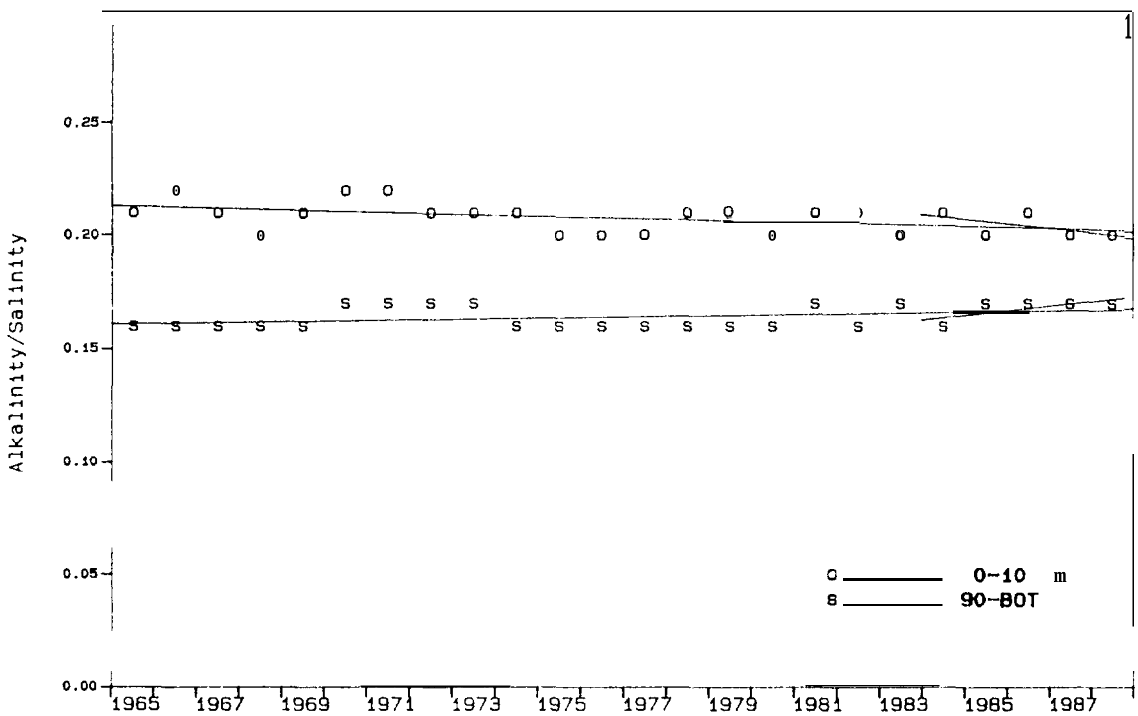


Figure 24. Long-term trends of annual means (A/S) in the surface water (0-10 m) and the bottom water (90 m to bottom) of the Western Gotland Basin from 1965 to 1988. Mean values of stations BY 34, BY 35 and BY 36 have been used.

Western Gotland Basin

Means for the stations BY 34, BY 35 and BY 36 have been used. Here the surface water shows a significant weak negative trend and the deep water (90 m - bottom) a weak positive trend, as in the northern Central Basin (Fig. 24 and Table 6).

2.3.3 Gulf of Bothnia

Åland Sea

A very weak positive trend could be found at the station F 64 (BMP D1) for the period 1965-1988 in the surface water, but there was not trend in the deep water (Table 6).

Bothnian Sea

Figure 25 represents A/S relations in the surface water and below 100 m in the Bothnian Sea. The vertical stratification in the water is very weak and therefore, the difference between the conditions in the surface and deep water is small and the regression lines for A/S are therefore very close to each other. The negative trend for the surface water is weak and not significant. For the deep water no trend can be found (Table 6).

Bothnian Bay

In the Bothnian Bay the conditions are very similar to the conditions in the Bothnian Sea (Fig. 26). In the surface water a negative trend can be found but no trend exists in the deep water (Table 6).

2.4 REGIONAL ASSESSMENT OF THE pH

The pH of the surface water shows seasonal variations due to the effects of primary phytoplankton production. In most cases annual means have been used here for trend calculations.

2.4.1 The Kattegat, the Sound and the Belt Sea

S. Fonselius[4] and H.P. Hansen'

Figure 27 shows the pH at the Kattegat station Fladen (BMP R6) as annual means in the surface water (0-10 m) and in the deep water (60 m - bottom) from 1965-1988. Both the surface and the deep water show a clearly positive trend, around 0.008 pH units/year (Table 7). The pH difference between surface and deep water is small, only around 0.1 pH units.

In the Sound the pH variations are shown at the station Landskrona Deep, 431 (BMP Q2). Figure 28 shows the variations at 0-10 m and 40 m - bottom from 1965-1988. In the surface water a very weak positive trend can be detected. In the deep water no trend could be found (Table 7). For the last 5 years both levels show a clear positive trend.

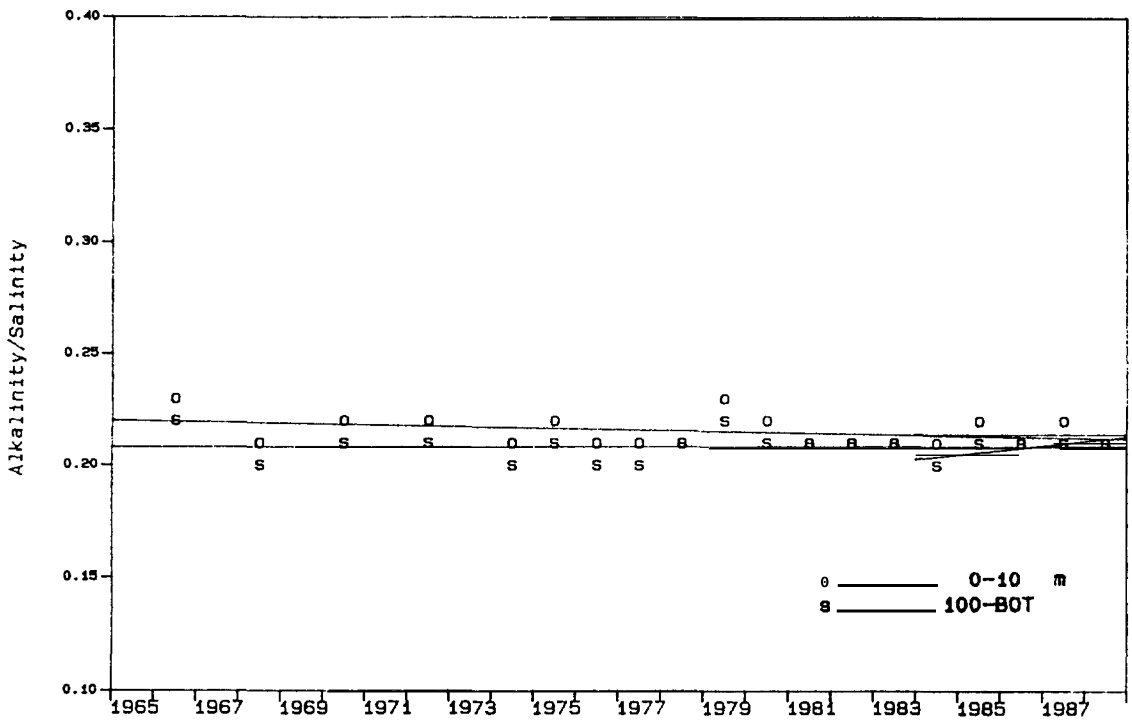


Figure 25. Long-term trends of annual means of A/S in the surface water (0-10 m) and the bottom water (100 m to bottom) of the Bothnian Sea from 1965 to 1988. Mean values of all deep stations have been used.

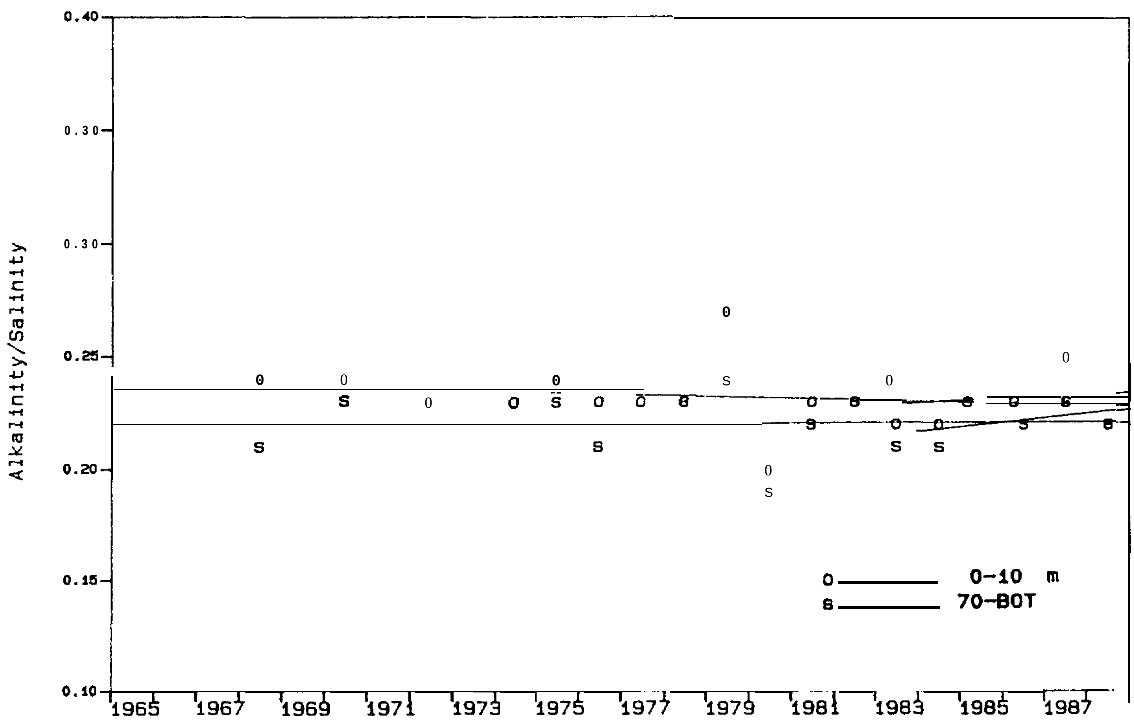


Figure 26. Long-term trends of annual means of A/S in the surface water (0-10 *m)* and the bottom water (70 m to bottom) of the Bothnian Bay from 1965 to 1988. Mean values of all deep stations have been used.

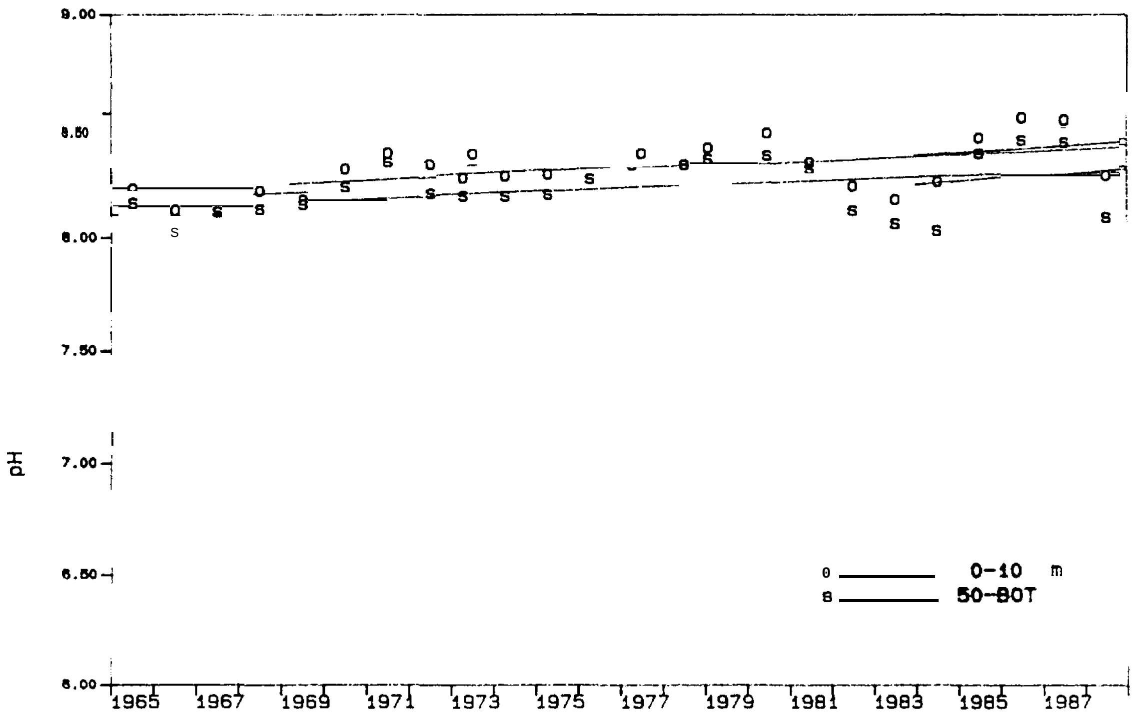


Figure 27. Long-term trends of annual means of pH in the surface water (0-10 m) and bottom water (50 m to bottom) of the Kattegat at station Fladen (BMP **R6)** from 1965 to 1988.

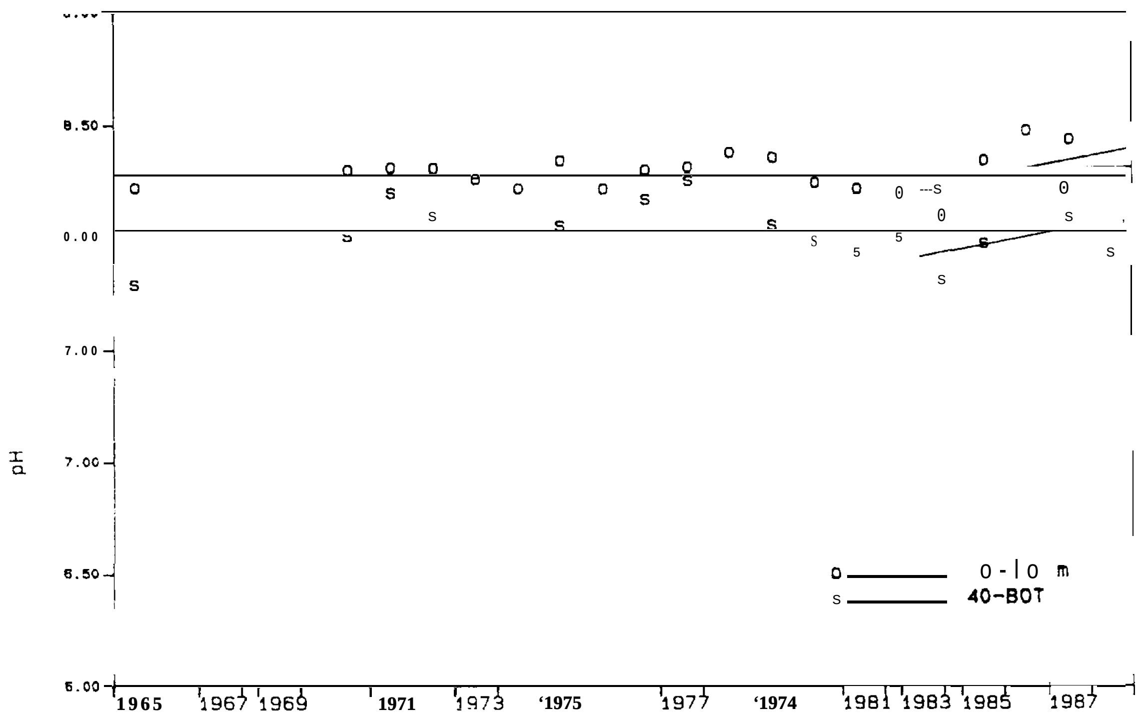


Figure 28. Long-term trends of annual means of pH in the surface water (0-10 m) and bottom water (40 m to bottom) of the Sound at the Landskrona Deep (BMP Q2) from 1965 to 1988.

Table 7. Long-term trends of annual means of pH in the Baltic Sea.

Area or station	Period	Depth m	Mean trend coefficient pH units/a	Overall trend pH units
Fladen	1965-1988	0-10	+0.008	+0.20
Fladen	1965-1988	60-bottom	+0.007	+0.18
Landskrona Deep	1965-1988	0-10	(+0.0025)	(+0.06)
Landskrona Deep	1965-1988	45-bottom	(0.000)	(0.00)
Arkona Deep	1965-1988	0-10	+0.008	+0.20
Arkona Deep	1965-1988	45-bottom	+0.008	+0.20
Bornholm Basin	1965-1988	0-10	+0.008	+0.20
Bornholm Basin	1965-1988	80-bottom	(+0.007)	(+0.16)
BY 8, 9	1965-1988	0-10	+0.008	+0.20
BY 8, 9	1965-1988	90-bottom	+0.017	+0.40
Gotland Deep	1965-1988	0-10	+0.007	+0.18
Gotland Deep	1965-1988	200-bottom	+0.009	+0.22
Gdańsk Deep	1969-1989	0-20	(+0.002)	(+0.04)
Gdaiisk Deep	1969-1989	100-108	+0.016	+0.34
BY 27, 28, 29	1965-1988	0-10	+0.008	+0.20
BY 27, 28, 29	1965-1988	125-bottom	+0.010	+0.24
Landsort Deep	1965-1988	0-10	+0.003	+0.08
Landsort Deep	1965-1988	400-bottom	+0.005	+0.12
BY 34, 35, 36	1965-1988	0-10	+0.008	+0.20
BY 34, 35, 36	1965-1988	90-bottom	+0.008	+0.20
Åland Sea	1965-1988	0-10	(+0.003)	(+0.08)
Åland Sea	1965-1988	200-bottom	+0.010	+0.25
Bothnian Sea	1965-1988	0-10	(0.000)	(0.00)
Bothnian Sea	1965-1988	100-bottom	+0.007	+0.18
Bothnian Bay	1965-1988	0-10	(0.000)	(0.00)
Bothnian Bay	1965-1988	70-bottom	+0.007	+0.16

The only significant trend of pH in the Kiel Bay is found in the bottom waters during the autumn minimum of oxygen (August-September; 25 m to bottom). The pH shows a clear negative trend of -0.03 units/a, which corresponds to the negative trend in oxygen concentrations (Fig. 29).

2.4.2 The Baltic Proper

Arkona Basin

S. Fonselius[4]

The Arkona Deep, BY 2 (BMP K4), represents the Arkona Basin. Annual means have been computed at 0-2 m and at 40 m - bottom from 1965-1988. The positive trends indicate an increase of pH both in the surface water and the deep water (Table 7).

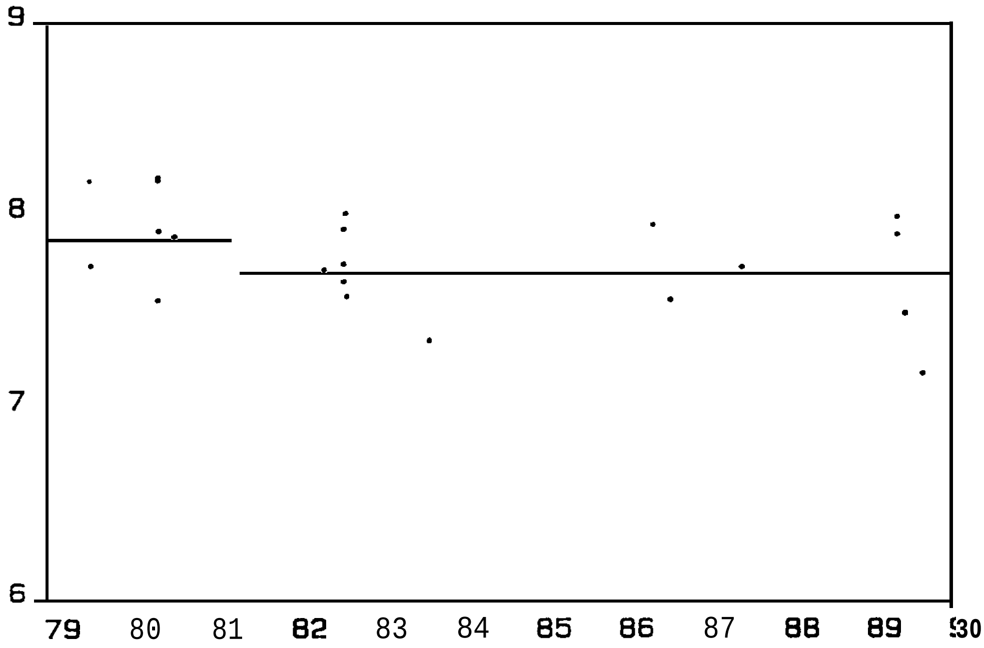


Figure 29. Long-term trend of pH in the bottom water (25 m to bottom) of the Kiel Bay during the autumn oxygen minimum (August-September) from 1979 to 1990.

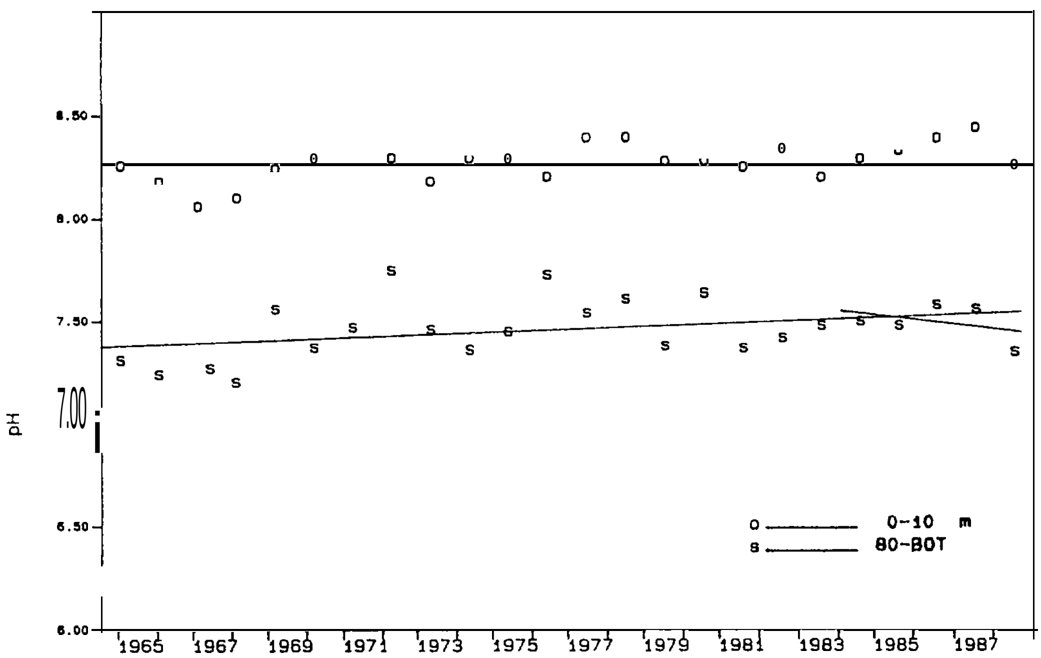


Figure 30. Long-term trends of annual means of pH in the surface water (0-10 m) and the bottom water (80 m to bottom) of the Bornholm Basin from 1965 to 1988. Mean values of stations BY 4 and BY 5 (BMP K2) have been used.

Bornholm Basin

S. Fonselius[4]

In the Bornholm Basin the means of the stations BY 4 and BY 5 **(BMP K2)** have been used. Figure 30 shows annual means at 0-10 m and 80 m - bottom from 1965-1988. Both plots show an increasing pH (approximately 0.008 pH units/year, Table 7). **In** the deep water the trend **becomes negative** for the 5 last years.

Eastern Gotland Basin

S. Fonselius[4]

In the Eastern Gotland Basin the stations BY 8 and BY 9 represent the southern part. **Figure** 31 shows the monthly means for the surface water (0-10 m) and for 90 m - bottom from 1965-1988. The positive trend for the surface water is approximately 0.008 pH units/year and for the deep water 0.017 pH units/year (Table 7). Figure 32 is from the Gotland Deep, BY 15 (BMP J1), and shows the trends at 0-10 m and 200 m - bottom as monthly means from 1965-1988. The increase is somewhat larger in the bottom layer, but both graphs show an increase of around 0.008 units/a (Table 7). The S-year trends 1984-1988 are opposite to similar trends at other stations.

Gdansk Basin

A. Trzosinskal

In the surface water of the Gdaiisk Deep (station P 1 = BMP L1) a significant increase in pH could only be found for the vegetative seasons of 1979-1986 (Fig. 33 **a, Table 8)** or **even** earlier (Table 2). Due to the sharp decrease of pH during the last two years (1987-1988) the overall trend for 1979-1988 became negative. In the deep and the intermediate water layers the increase of pH had been observed since 1969 but during the last decade the rate of increase was slower (Fig. 33 b, Table 8). The positive trend of pH at 80 m depth corresponds well with the long-term increase of oxygen concentration in the intermediate water layers (Table 5, Fig. 13 b); in the bottom water both parameters show opposite trends.

Table 8. Long-term variations of pH in the Gdansk Deep (P 1 = BMP L1).

Period	Depth m	Mean trend coeff. pH units/a	Overall trend pH units	Remarks
1979-1986	0-20	0.028	0.22	April-Ott
1979-1988	0-20	-0.019	-0.19	April-Ott
1969-1988	80	0.019	0.38	
1969-1989	100-108	0.016	0.34	
1979-1989	100-108	0.013	0.14	

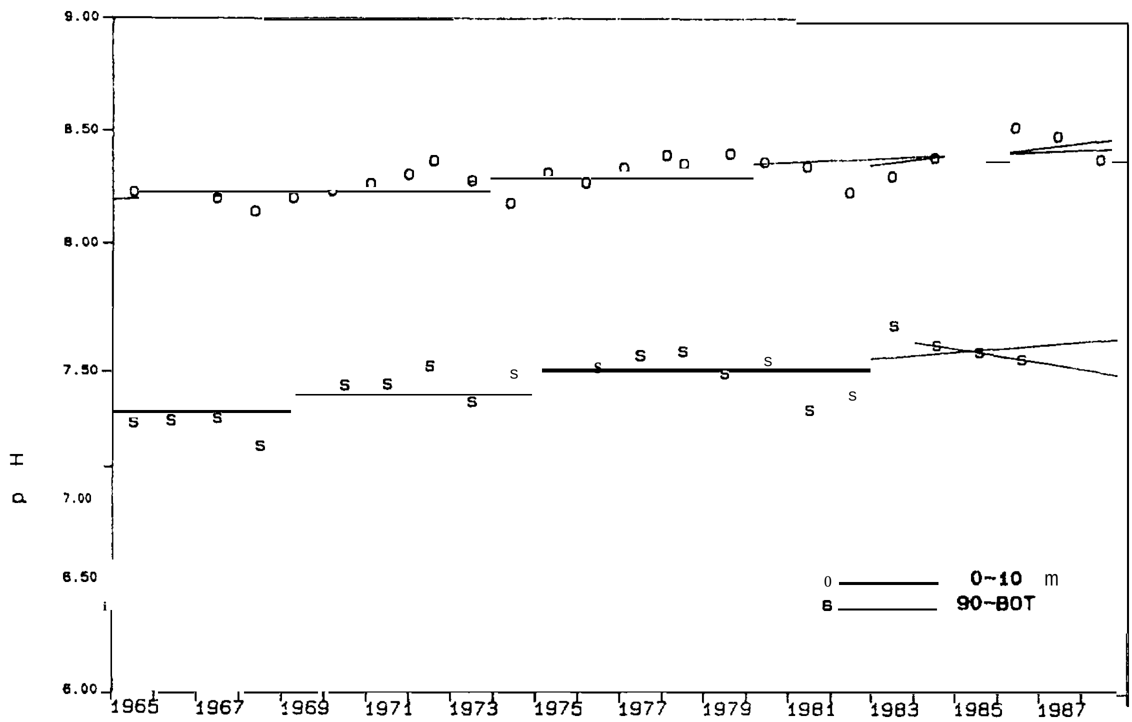


Figure 31. Long-term trends of annual means of pH in the surface water (0-10 m) and the bottom water (90 m to bottom) of the southern part of the Eastern Gotland Basin from 1965 to 1988. Mean values of stations BY 8 and BY 9 have been used.

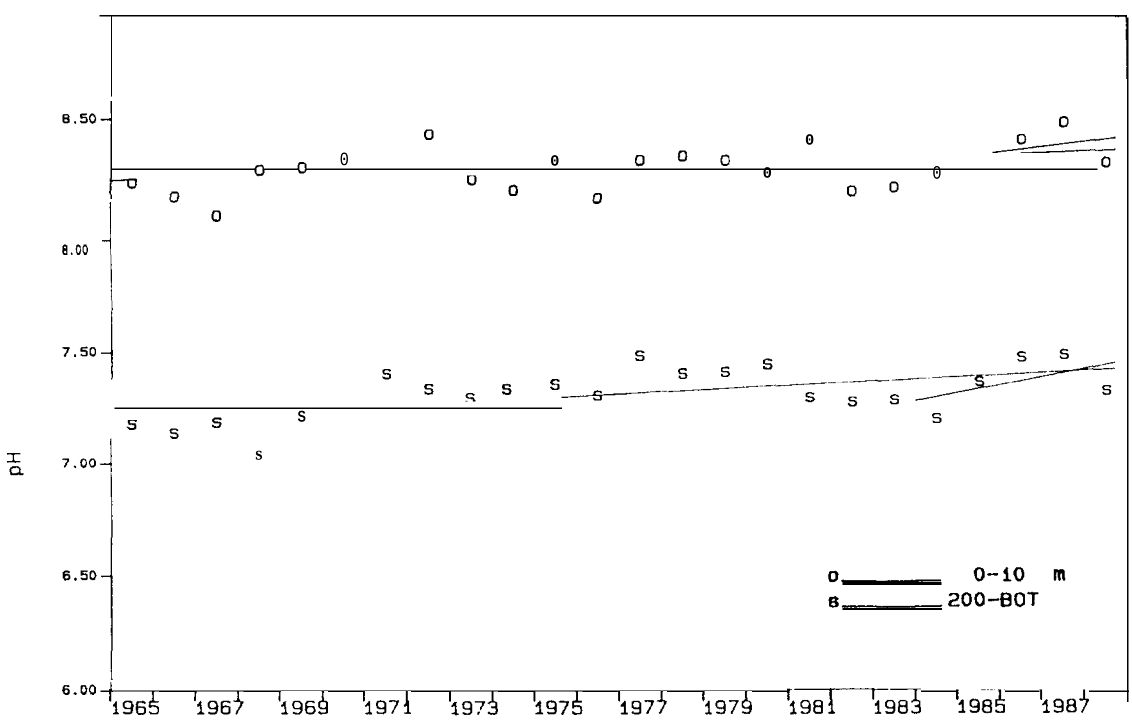
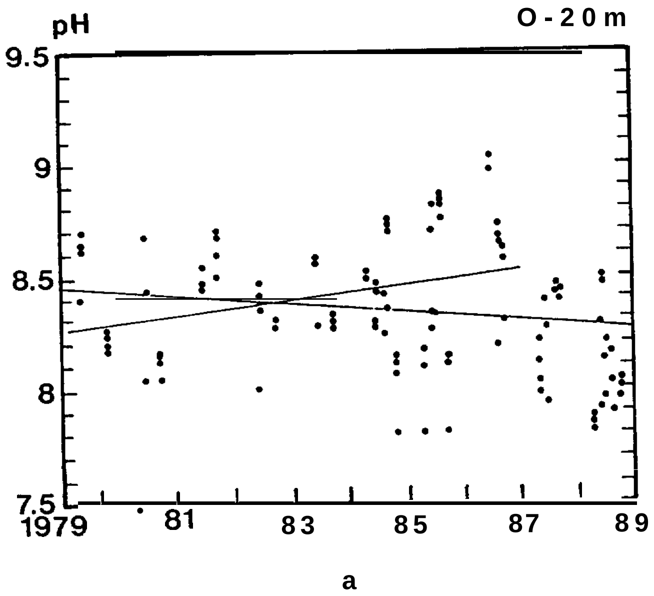


Figure 32. Long-term trends of annual means of pH in the surface water (0-10 m) and the bottom water (200 m to bottom) of the Gotland Deep, BY 15 (BMP Jl), from 1965 to 1988.



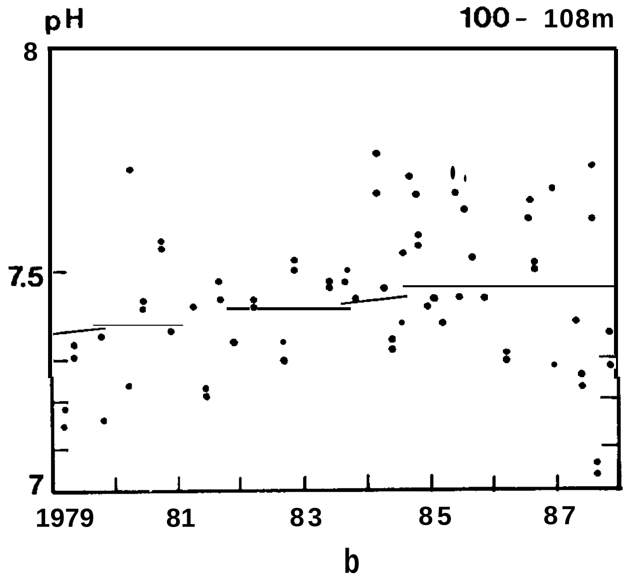


Figure 33 a. Long-term variations of pH in the 0-20 m surface water in the Gdańsk Deep, P 1 (BMP L1), during the warm seasons 1979-1988.
b. Long-term variations of pH in the 100-108 m bottom water in the Gdańsk Deep, P 1 (BMP L1), from 1979 to 1988.

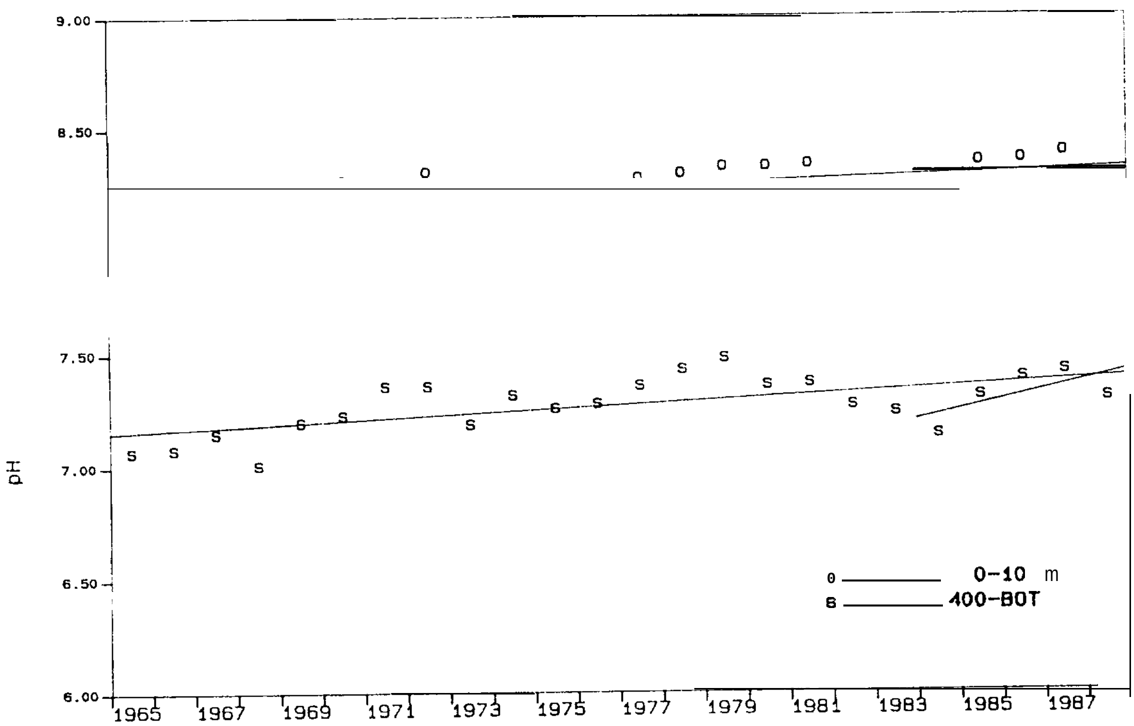


Figure 34. Long-term trends of annual means of pH in the surface water (0-10 m) and the bottom water (400 m to bottom) of the Landsort Deep, BY 31 (BMP H3), from 1965 to 1988.

Northern Baltic Proper
S. Fonselius[4]

Here the pH means of the stations BY 27, BY 28 (BMP H2) and BY 29 have been used for the central part. In Table 7 annual mean trend coefficients at O-10 m and 125 m - bottom, respectively, for the period 1965-1988 are given. The trend shows an increase of pH both in the surface and the bottom water. The Landsort Deep; BY 31 (BMP H3), represents the western part of the basin. Figure 34 shows the annual mean trends for O-10 m and 400 m - bottom. Also here the trend shows a clear pH increase in both layers (Table 7).

Western Gotland Basin
S. Fonselius[4]

Annual means of the stations BY 34, BY 35 and BY 36 at O-10 m and 90 m - bottom are shown in Figure 35. A clear positive trend in pH can be seen (Table 7).

2.4.3 Gulf of Bothnia
S. Fonselius[4]

Åland Sea

In Table 7 annual means at the station F 64 (BMP D1) at O-10 m and 200 m - bottom are given from 1965-1988. In the surface water only a very weak positive trend in pH can be found, but in the deep water the pH trend is clearly positive and highly significant. For the last 5 years a decreasing trend could be seen.

Bothnian Sea

Figure 36 shows annual pH means of a mean of the representative deep stations in the Bothnian Sea at O-5 m and 100 m - bottom from 1965-1988. In the surface water no trend can be seen, but in the deep water the trend is clearly positive (Table 7).

Bothnian Bay

Figure 37 finally shows the pH trends at O-10 m and 70 m - bottom as annual mans for a mean of representative deep stations. In the surface water no trend can be found, but in the deep water the trend is positive for the period 1965-1988 as in the Bothnian Sea (Table 7).

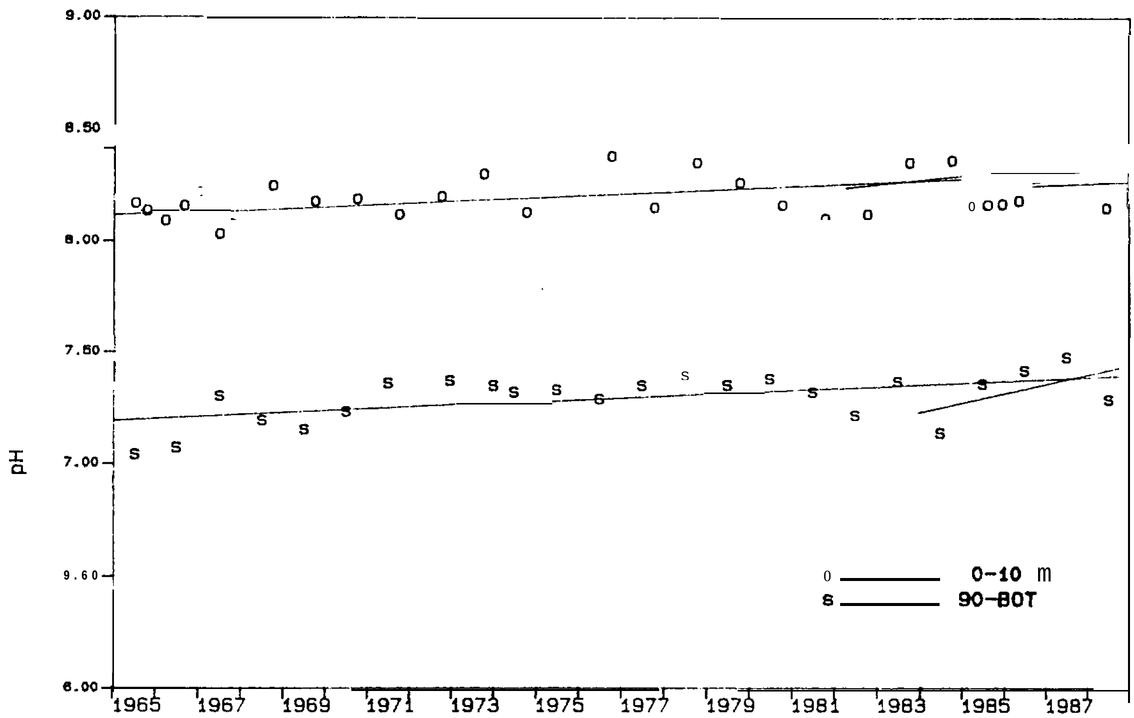


Figure 35. Long-term trends of annual means of pH in the surface water (0-10 m) and the bottom water (90 m to bottom) of the Western **Gotland** Basin from 1965 to 1988. Mean values of stations BY 34, BY 35 and BY 36 have been used.

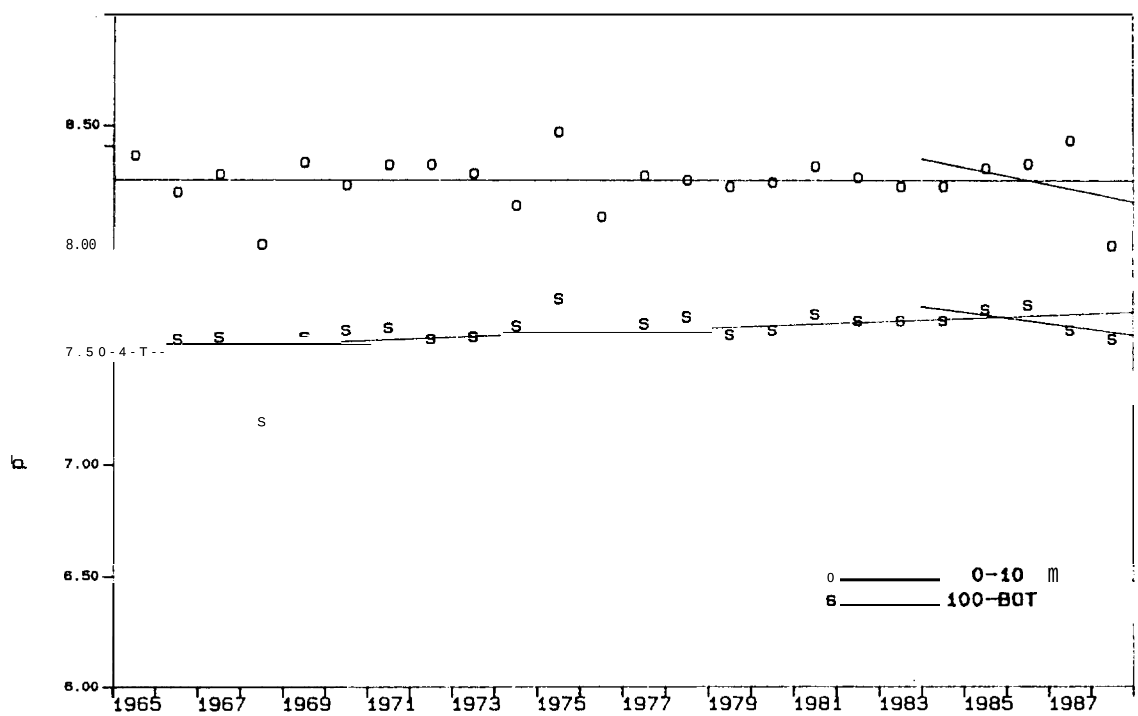


Figure 36. **Long-term trends of annual means of** pH in the surface water (0-10 m) and the bottom water (100 m to bottom) of the Bothnian **Sea** from 1965 to 1988. **Mean values** of all deep stations have been used.

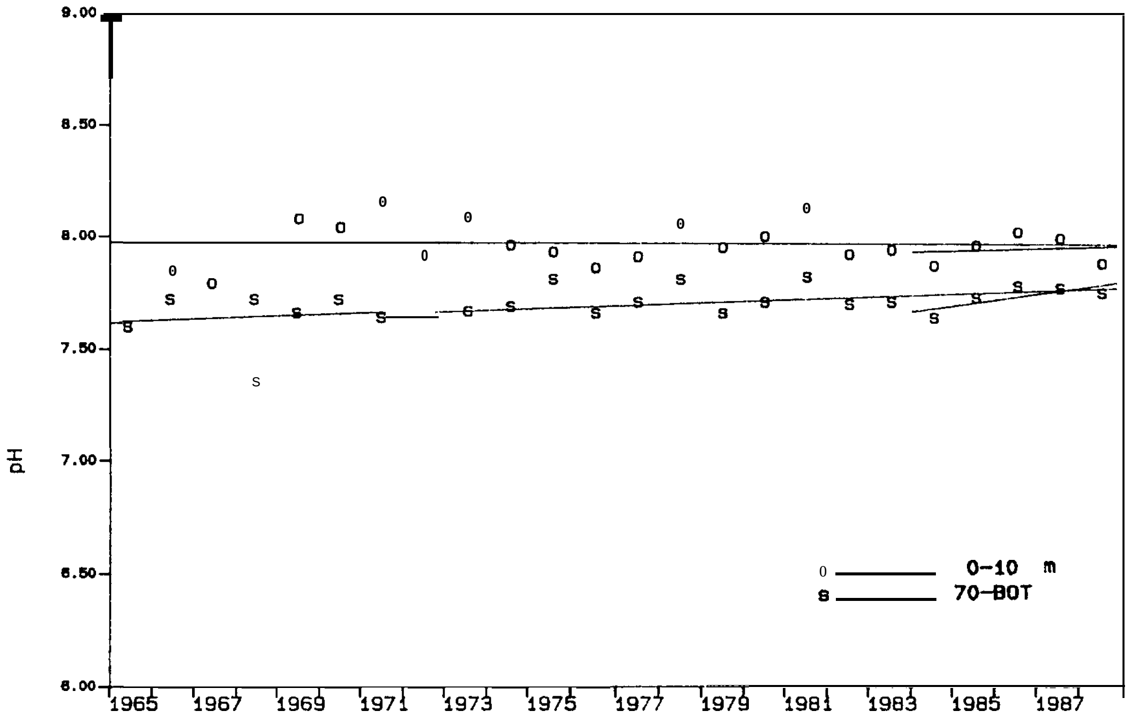


Figure 37. Long-term trends of annual means of pH in the surface water (0-10 m) and the bottom water (70 m to bottom) of the Bothnian Bay from 1965 to 1988. Mean values of all deep stations have been used.

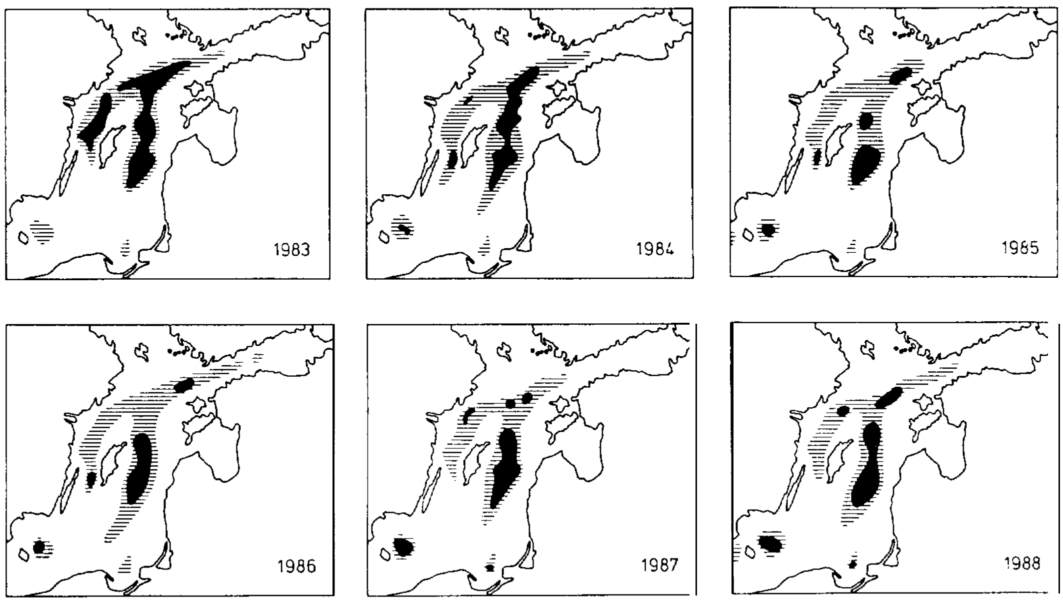


Figure 38. Occurrence of hydrogen sulphide (black areas) in the deep basins of the Baltic Proper between 1983 and 1988 (after Nehring and Francke, completed after Andersin and Sandler 1989). Shaded areas show oxygen concentrations below 2 ml/l. The maps give maximum coverage of the observed areas during each year.

2.5 SUMMARY

S. Fonselius[4] and A. Trzosiiiska[1]

2.5.1 Oxygen

In the intermediate water of the Eastern Gotland Basin and the Northern Baltic Proper, and in the deep water of the Western Gotland Basin the oxygen trend is positive due to inflow of new water in the layers below the halocline and also due to the sinking of the halocline in the Eastern Gotland Basin. In the bottom water of the Kattegat, the Arkona Basin, the Bornholm Basin and the Eastern Gotland Basin the oxygen trend is negative due to lack of effective renewal of the bottom water and also due to increased eutrophication of the surface water. In the bottom water of the Northern Baltic Proper no trend can be found. The water has lost almost all its oxygen, but only very small amounts of hydrogen sulphide have occasionally been formed. The bottom water of the Åland Sea shows a weak positive trend, but in the Bothnian Sea and the Bothnian Bay the trends are negative (Table 3). The 5-year period (1984-1988) is too short for any definite conclusions on the direction and the rate of oxygen trends.

Studies carried out by Andersin and Sandler (1989) on the occurrence of hydrogen sulphide and low oxygen concentrations in the bottom waters of the Baltic Proper and the Gulf of Finland (Fig. 38) show that the total area of sea bed covered by waters with unfavourable oxygen conditions decreased significantly during the period 1963-1987, while areas covered by hydrogen sulphide containing water showed a weak, statistically non-significant positive trend. They concluded that the bottom areas suitable for benthic fauna have not diminished in the Central Baltic Proper and the Gulf of Finland during the last 25 years. A clear spreading has taken place in the Arkona Basin and the Bornholm Basin because the duration of periods with favourable oxygen conditions have become shorter.

The main reason for the decreasing oxygen concentrations in the bottom water of the deep basins is obviously the long stagnation period. No major inflows of Kattegat water have occurred since 1977. This can be seen from the continuously decreasing density of the bottom water, e.g. in the Eastern Gotland Basin. It is quite clear that this kind of stagnation is a natural phenomenon associated with river run-off and meteorological conditions. Another effect of this stagnation is that the halocline is moving downward. This increases the supply of oxygen to the intermediate levels (Chapter 1 "Hydrography").

2.5.2 Alkalinity

The trends are mostly negative in the surface water but a positive trend can be found in the Åland Sea. In the Bornholm Basin and the Eastern Gotland Basin no trend can be found in the bottom water. In the Northern Baltic Proper and the Western Gotland Basin the trend in the bottom water becomes positive, but it is very weak. In the Gulf of Bothnia and the Åland Sea no trend can be found in the bottom water. When the results in Table 6 are compared with the results in Table 1, it can be seen that the annual mean variations of the specific alkalinity A/S, and in many cases even the overall 24-year trends (1965-1988) are smaller than the standard deviations of A/S mean values found by different authors in the surface and the bottom waters. One may conclude that the long-term variations of the A/S ratio might be inside the analytical errors, the more so as the majority of presented trends are statistically non-significant. The 5-year trends for 1984-1988 are too short for any definite conclusions.

Alkalinity is strongly related to the salinity and is nearly a conservative parameter. In the surface water there are signs of a very weak negative trend for the relation A/S, but it is hardly significant. The deep water generally has a constant A/S relation. For calculation of total carbonate in the Baltic Sea the alkalinity can be calculated from Buch (1951). For more exact work it is still necessary to determine the alkalinity in samples of sea water in the Baltic Sea. The weak negative surface trend has to be investigated more closely. Decreasing salinity in the Baltic Sea should result in **an increase** of the alkalinity, because the river water has a higher alkalinity. According to Chapter 1 "Hydrography", the salinity in the Baltic Proper is decreasing, but the alkalinity is also decreasing in the surface water and is unchanged in the deep water. Therefore, the very small effect of the decreasing salinity is masked by some more important processes. The explanation may be the increasing eutrophication in the Baltic Sea, which increases the primary production of organic matter. This leads to a decrease of total carbonate and an increase of pH.

2.5.3 pH

Most stations and sea areas show clearly positive trends both in the surface water and the deep water. In the Sound (Landskrona Deep) no trend can be found in the bottom water. In the Rothnian Sea and the Bothnian Bay no long-term trends for the surface water can be detected. In the Kiel Bay the pH trend is negative. Also here the 5-year trends for 1984-1988 are too short for conclusions. Table 7 shows the long-term pH trends.

On annual basis there seems to be a positive trend of the pH in the Baltic water, both in the surface and the deep water. The reason for this is not clear. One would expect that the pH of the surface water should increase due to the increasing eutrophication and therefore increasing primary production. For the deep water the most probable explanation is that increasing production and sedimentation of organic matter, followed by increasing oxygen consumption for its decomposition create conditions for denitrification. During the nitrate reduction OH^- ions are produced and ammonia is accumulated in stagnant anoxic waters. There is also another mechanism which should be taken into account when considering long-term variations of pH in the Baltic waters. Acid rains may enhance weathering of carbonate rocks resulting in escape of carbon dioxide into the atmosphere and in release of oxides of the alkaline earth metals into the water.

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3. NUTRIENTS

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ABSTRACT

Concentrations of phosphate, total phosphorus, nitrate, total inorganic nitrogen compounds and silicate were studied with respect to trends in the surface layer and in the deep water.

Since the late 1970s concentrations of the phosphorus and nitrogen compounds are no longer increasing, on average, or are characterized by lower accumulation rates with respect to the previous period in the surface layer in winter in most parts of the Baltic Sea Area, whereas silicate concentrations generally decrease in this layer. The strong increase of phosphate concentrations identified in the near-bottom water layers of the central Baltic basins since 1977 mainly results from transient phosphate remobilization from the iron hydroxo complex in the sediments caused by the increasing hydrogen sulphide concentrations in the recent stagnation period.

3.1 INTRODUCTION AND GENERAL REMARKS

D. Nehring[1]

One of the most important results of the assessments 1980 and 1980-1985 was the evidence of eutrophication in the Baltic Sea Area during recent decades (Baltic Marine Environment Protection Commission, 1981, 1987a). The winter pool of nutrients in the surface layer has increased with respect to nitrate or inorganic nitrogen compounds (NO_3+NO_2+NH_4) in all Baltic subregions. The concentrations of phosphate or total phosphorus also increased, on average, except in the Bothnian Bay and the Gulf of Riga. The statistically significant overall trends of the phosphorus and nitrogen compounds were interrupted by a period with decreasing or nearly constant concentrations in the Baltic Proper, the Belt Sea, and the Kattegat between 1976 and 1980. The phosphate and nitrate variations observed in winter were closely correlated with the salinity (density) in the Baltic Proper, indicating that eutrophication in this area is also affected by the advective deep water exchange remobilizing nutrients into the surface layer.

As in the surface layer, positive trends of phosphorus and nitrogen compounds were also identified below the halocline. Denitrification during anoxic transition and phosphate liberation from sediments in the presence of hydrogen sulphide made it difficult to identify long-term trends in some of the Baltic deep basins.

Regardless of whether anthropogenic activities or natural variations play the dominating part in the nutrient increase, eutrophication was identified in the assessment 1980-1985 as one of the most serious problems in the Baltic Sea area (Baltic Marine Environment Protection Commission, 1987a). The main negative changes in the marine environment were those concerning trends towards increasing nutrient concentrations which cause a higher biological productivity. The organic matter produced in this process consumes oxygen during its microbial destruction, thus contributing to the more frequent oxygen depletion and the occurrence and spreading of hydrogen sulphide in the bottom water layer of the Baltic Proper, the Belt Sea and the Kattegat under stagnant conditions.

Nutrients discussed in the present assessment are the inorganic nitrogen compounds and phosphate limiting primary production. Silicate, although being of lower significance in this connection, is also included in the investigations.

Total phosphorus and total nitrogen analyzed by means of persulphate digestion are studied as well. The results from the total nitrogen analysis should be looked upon with caution since possibly not all nitrogen fixed in organic substances like humic acids is oxidised by this method.

The results of nutrient intercalibrations and the comparability of nutrient data have been already discussed in the earlier assessments (Baltic Marine Environment Protection Commission, 1981, 1987a). Nutrient data are available from the HELCOM Data Base storing data of the Baltic Monitoring Programme of HELCOM. Additional data results from national monitoring activities.

3.2 NEW RESULTS WITH RESPECT TO *CHANGES* IN THE BALTIC ECOSYSTEM

D. Nehring[1]

Simulations performed by Wulff and Rahm (1989) show that a remarkably small number of stations is sufficient for the calculation of large-scale and long-term variations in the Baltic Sea. Since the seasonal changes in nutrient concentrations are often more substantial, the increase in observation frequency seems more appropriate for the improvement of the Baltic Monitoring Programme than the excessive geographical resolution.

Voipio and Tervo (1988) studied the influence of the sampling dates on nutrient trends in the surface layer of the Gulf of Finland. Since the nutrient level increases in this layer during the late autumn and the early winter, a delay in the sampling dates generates trends. For this reason, trend studies should be performed in the surface layer only in the later winter, when the nutrient pool is fully developed. These findings give rise to reconsider earlier conclusions (Baltic Marine Environment Protection Commission, 1987a) concerning long-term trends of nutrients in the Gulf of Finland.

Baltic monitoring data were reviewed by Wulff and Rahm (1988) with respect to long-term, seasonal, and spatial variations of nutrients. In contrast to the storage of nitrogen and phosphorus compounds, the silicate pool is decreasing in the Baltic Sea in recent decades. The spring bloom intensified by eutrophication and a subsequent sedimentation of diatoms are discussed in this connection. Although silicate is only considered a limiting factor in the Kattegat and the Belt Sea, this nutrient requires more attention in future monitoring activities in the Baltic Sea.

Investigations on long-term variations of phosphorus and nitrogen compounds performed by Yurkovskis (1987) in the Eastern Gotland Sea confirm the results of the assessment 1980-1985 (Baltic Marine Environment Protection Commission, 1987a). The increasing phosphate and nitrate concentrations in the range of the halocline were attributed to the anthropogenic impact. The author also studied the nutrient dynamics in relation to the hydrographic conditions, the oxygen regime and the biogeochemical cycle. In this connection, Yurkovskis and Kaleis (1987) related the river run-off with the salinity and the phosphorus dynamics. Increasing river run-off weakens the inflow of saline water through the Danish Straits and decreases the salinity in the surface layer of the Central Baltic Proper with a delay of 4-5 years. Independent from the river discharge, the salinity recently showed stronger decreases below the halocline as in the surface layer. The river run-off characterized by positive anomalies since the middle of the 1970s fluctuates in periods of 7-16 years. The decrease of the density gradient varying in time favours the vertical exchange through the halocline and the phosphate accumulation in the surface layer during the winter. The dynamics of phosphorus depend on different factors complicating the understanding of the year-to-year variations of this nutrient in the Baltic Sea.

Yurkovskis (1989) extended the description of phosphate, total phosphorus, nitrate and ammonium trends in the central Baltic Sea now covering the period 1961-1985. His results agree well with recent investigations by Nehring and Matthäus (1990) indicating positive overall trends in the phosphate and nitrate concentrations, which are characterized by decreasing accumulation rates since the later 1970s with the anoxic deep waters as an exception.

Cyberska et al. (1987) analyzed the nutrient situation in the Bay of Gdańsk between 1974 and 1983. They identified positive phosphate and nitrate trends in the surface layer in winter as well as in the deep water. Interannual and long-term variations were related to changes in the hydrographic conditions and the discharge of the Vistula River.

Trzosifiska (1990) studied time series of phosphate (1961-1984) and nitrate (1969-1984) concentrations from the Gdańsk Deep by means of spectral correlation analysis and defined oscillations which took place independently of the eutrophication. The spectral power functions indicated that the phosphate concentrations oscillated in cycles of 3, 6 to 7, and 10 to 12 years in the winter surface water and at 80 m and 100 m depths. Periods of 3 to 4 and 5 to 7 years were identified for nitrate. The 3 to 4 and the 6 to 7 year cycles were attributed to changes in the atmospheric circulation system creating salinity fluctuations due to variations in the river outflows and the oceanic inflows, while the periods of 9 to 12 years seem to be exerted by the periodicity of the sun spots activity.

Studies carried out by Nehring (1989) in the bottom water of the Gotland Deep yield a negative trend in the phosphate concentrations. That was attributed to the decreasing pool of phosphate which can be remobilized under anoxic conditions from the sediments or its interstitial waters (cf. Yurkovskis, 1987). The trend became positive and was of the same order as in the 100 m depth when phosphate concentrations measured in the presence of hydrogen sulphide were ignored. Positive nitrate and phosphate trends were also identified in the deep water of the Bornholm Basin, when values influenced by denitrification or anoxia were excluded.

The ratio of oxygen consumption to phosphate accumulation was not well defined in Central Baltic Proper deep waters due to phosphate remobilization from the sediments under anoxic conditions (Nehring, 1989). On the other hand, Shaffer (1987) found a nearly constant ratio of 318:1 by mole in the well oxygenated isopycnal surface above the Baltic redoxcline. Earlier investigations by Majeweski et al. (1976) have shown that the average ratio of oxygen consumption to phosphate regeneration due to organic matter decomposition was not very much higher in the well oxygenated Baltic water (286:1) than the Redfield ratio (276:1). But there were remarkable regional differences. The mean oxygen equivalent found for the oxygen depleted Baltic deep waters was much lower (74:1), demonstrating high efficiency of bottom sediments as an additional source of phosphate. The northward increase of this equivalent ratio implied that this source was more effective in the southern than in the northern areas of the Baltic Proper.

Fonselius et al. (1988) observed a patchy distribution of nutrients during the spring bloom of the phytoplankton in the Central Baltic Proper. The algae consumed nitrate and phosphate in the Redfield ratio (16:1, by mols), although the respective nutrient ratio in the Baltic surface layer is significantly lower during the winter.

Baltic Marine Environment Protection Commission (1987b) prepared the first pollution load compilation of the Baltic Sea using the results of a questionnaire distributed to the Baltic countries. The total annual input of nutrients from the drainage area amounted to 528 000 t nitrogen and 49 000 t phosphorus. This is only a rough estimation, since the information and data from the separate countries are not strictly comparable and show gaps.

Rönner (1985) studied the nitrogen transformation in the Baltic Proper. His budget accounts for a nitrogen loss by denitrification of 470 000 t/a below the halocline amounting to roughly 80% of the municipal and industrial discharges by the river run-off. This result shows that denitrification counteracts eutrophication.

The residence times of nutrients and humic substances were studied by Wulff and Stigebrandt (1989). The overall budgets calculated in this connection show that about 90% of the nutrient losses are due to biogeochemical sinks within the Baltic Sea. This means that perturbations in the water exchange with the North Sea should have little effect on the nutrient budgets of the Baltic ecosystem. Utilizing the best available estimates of the supplies as well as the storage changes and considering the advective transfer between the basins, the authors have been able to determine the magnitude of the internal biogeochemical sinks of total phosphorus (37 000 t/a), total nitrogen (880 000 t/a), and silicate (700 000 t/a) in the Baltic Sea as a whole as well as in its subregions. A simple time dependent model for the winter surface concentrations of total nitrogen and phosphorus in the Baltic Proper was run with estimated loading for the period 1950-2000, supplementing the model for the dynamics of nutrients and oxygen in Central Baltic Proper deep waters (Stigebrandt and Wulff, 1987).

In other simple models with empirical mathematical descriptions, Schrsder and Malmgren-Hansen (1988) related the decline of the oxygen concentrations below the halocline with the four-fold increase in the nitrogen load of the Baltic Proper during the present century.

Savchuk (1987) and Savchuk et al. (1989) developed a box model of long-term dynamics of matter which was used to hind cast year-to-year variations of organic matter, inorganic nitrogen and phosphorus compounds and oxygen in the Baltic Sea for the period 1951 to 1982. The numerical experiments let the authors **suggest** that the most important reason of the eutrophication in the Baltic Sea is the increase in anthropogenic loads of organic matter and nutrients, while natural changes cause only background year-to-year variations.

Increasing attention is paid to urea in the Baltic Sea environment. Although its average concentrations are low in different areas and depths, ranging only between 0.3 and 0.5 $\mu mol/dm^3$, Irmisch (1986) and Valderrama (1987) identified seasonal variations, indicating that urea is linked with the biochemical cycle of nitrogen. Irmisch (1989) proved that phytoplankton is much more active as bacteria in the transformation of urea labelled with radiocarbon.

3.3 REGIONAL ASSESSMENTS OF LONG-TERM VARIATIONS

D. Nehring[1]

The data from the Baltic Monitoring Programme (BMP) are stored in the HELCOM Data Base. Using these data the HELCOM data consultant, Prof. M. Perttilä, FIMR, prepared figures of the nutrient distribution at BMP stations for distinct depths, layers and seasons. An example concerning phosphate time-series at BMP stations in the most important sub-regions of the Baltic Sea area is shown in Figure 1. Although phosphate is one of the best studied parameters in the BMP, its distribution is often characterized by gaps mainly in the surface layer in winter but also in the year-round data in the deep water. We get similar or still more insufficient figures for the distribution of total phosphorus, nitrate, total inorganic nitrogen, total nitrogen, and silicate. These gaps inhibit trend studies. Additionally, the time series available from the HELCOM Data Base covering only the period 1979-1988 are not long enough to study long-term variations. Therefore, the following trend assessments also include national nutrient data.

Trends are studied by linear regression analysis. Obvious outliers were not included in the investigations and are enclosed in brackets in the figures. This is also true for mean annual variation rates (trend coefficients) with $p>0.05$ (Student's t-test) in the tables.

Separation between the layer above and below the permanent halocline is often important for trend analysis in the Baltic Sea Area since oceanological conditions are quite different in these water bodies. Nutrient concentrations in the surface layer fluctuate seasonally because they are linked into the biological cycle. Trend studies of inorganic nutrients in this layer are therefore meaningful only in winter and early spring when phytoplankton production is limited by low light intensity and nutrient mineralization and seasonal accumulation has finished. The period suitable for these investigations varies since phytoplankton development starts earlier in the western than in the central or northern parts of the Baltic Sea (Kaiser and Schulz, 1989), whereas nutrient accumulation is generally finished in the beginning of January. Trend studies of total phosphorus and nitrogen do not depend so much from the season.

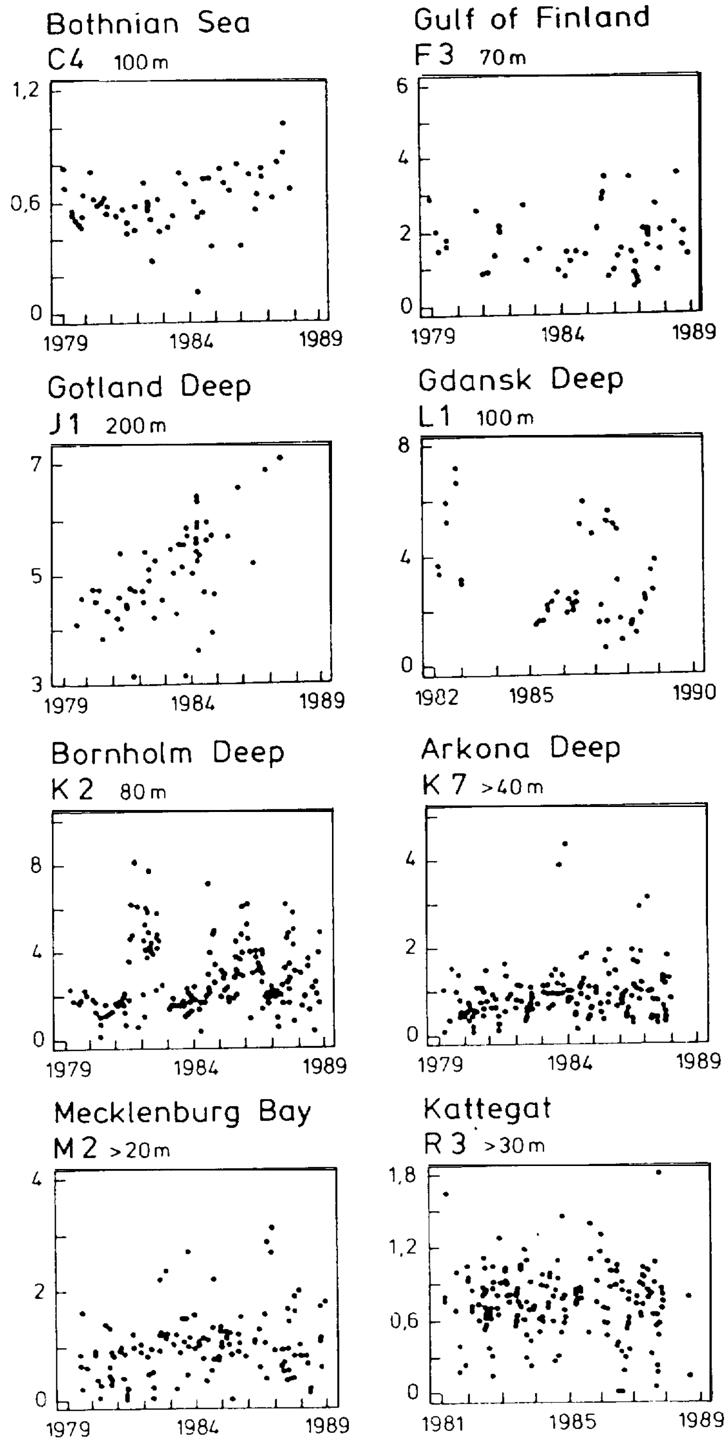

Figure 1. Phosphate time-series in deep waters at BMP stations in the most important Baltic sub-regions from data stored in the HELCOM Data base (concentrations in μmol dm^{-3}). Station C4 = SR 5; F3 = LL7; J1 = BY 15; L1 = P1; K2 = BY 5; K7 = BY 1; M2 = Bay of Mecklenburg; R3 = 413.

Oxygen conditions have a strong influence on the distribution of inorganic phosphorus and nitrogen compounds in the deep water. Nitrate is denitrified in the course of the transition from oxic to anoxic conditions. Nitrate trend estimations are therefore impossible or of restricted reliability in the deep water where conditions are sometimes anoxic. In addition, if hydrogen sulphide is present, large amounts of phosphate are liberated from the sediments. Phosphate concentrations therefore rise steeply under these conditions. Moreover, ammonium is the final product of nitrogen mineralization and is also remobilized from the sediments under anoxic conditions. These connections, which have been discussed comprehensively among others by Nehring (1987, 1989), must be taken into account, when phosphate and nitrate trends are studied in central Baltic deep waters.

The hydrographic subdivisions of the Baltic Sea area and the BMP stations used in the following regional assessments are shown in the maps of the introductory chapter.

3.3.1 **The Kattegat**

L.A. Jorgensen[3]

Data collected from the Danish National Environmental Research Institute in the period 1974-1988 *were* used for trend studies in the Kattegat. The winter concentrations include all data between the beginning of January and the middle of February. Data up to the end of March are added as long as the phytoplankton spring bloom had not developed.

Tables 1 and 2 contain the mean annual accumulation rates of totai phosphorus, total inorganic nitrogen compounds (NO_3 + NO_2 + NH_4), and silicate in the Kattegat according to the periods under investigation. The rates for nitrogen under winter conditions are very similar for both the surface layer and the whole water column. Nitrogen concentrations show low accumulation rates in the deep water considering year-round data.

Trend studies for different periods yield negative nitrogen trends in the beginning and positive nitrogen trends since about 1982. This supports the conclusion by Ertebjerg et al. (1981) that eutrophication was not a problem in the Kattegat area in the 1970s.

Accumulation rates of the inorganic nitrogen compounds covering the period 1983-1988 are characterized by strong variations between the different stations (Table 2). The mean concentrations for the period 1974-1982 are, however, lower than those of the last period.

Total phosphorus shows also increasing winter concentrations (Table 2). The overall trends are positive at all stations in the Kattegat.

In contrast to phosphorus and nitrogen, winter concentrations of silicate decrease in the period under investigation reaching high negative rates (Table 2) in the whole Kattegat area. The results for the surface layer and for the whole water column differ insignificantly.

Table 1. Mean annual accumulation rates (trend coefficients) of the inorganic nitrogen compounds ($NO3 + NO2 + NH_4$) in the Kattegat. Period under investigation 1974 – 1988.

BMP Stations	Surface layer (O-10 m) winter data only $\mu mol\ dm^{-3}\ yr^{-1}$	Deepwater (20m – bottom) year-round data $\mu mol\ dm^{-3}\ yr^{-1}$
R 1 (Gniben)	0.42	(0.12)
R 2 (Kullen)	(0.35)	0.01
R 3 (Anholt E)	0.30	0.05
R 4 (Aalborg B.)	0.33	
R 6 (Fladen)	(0.35)	0.07
Mean	0.35	0.06

Table 2. Mean annual accumulation rates (trend coefficients) of total phosphorus, inorganic nitrogen compounds ($NO_3 + NO_2 + NH_4$), and silicate in the Kattegat in winter. Whole water column.

BMP Stations	tot. P $\mu mol\ dm^{-3}\ yr^{-1}$ 1974-1988	$NO_3+NO_2+NH_4$ $\mu mol\ dm^{-3}\ yr^{-1}$ 1974-1988	1983-1988	SiO_4 $\mu mol\ dm^{-3}\ yr^{-1}$ 1974-1988
R 1 (Gniben)	(0.032)	(0.41)	-0.08	(-0.65)
R 2 (Kullen)	(0.035)	(0.38)	(0.54)	(-0.64)
R 3 (Anholt E)	(0.038)	(0.23)	0.24	(-0.49)
R 4 (Aalborg B.)	(0.048)	(0.38)	0.77	(-0.40)
R 6 (Fladen)	(0.055)	0.35	(2.07)	(-0.55)
Mean	0.043	0.35	(0.74)	-0.55

The accumulation rates of total phosphorus and inorganic nitrogen compounds are nearly the same as in earlier studies in the Kattegat comparing the values of the surface layer in winter (Baltic Marine Environment Protection Commission, 1987a). In contrast to the upper layer, recent investigations yield much lower rates for nitrogen accumulation in the deep water.

Long-term variations of phosphate and nitrate covering the period 1964-1987 were studied by Engström and Fonselius (1989) using data of the Kattegat Station Fladen (BMP R 6). Taking only values from the winter months, the concentrations of both nutrients are significantly increasing in the surface layer (Fig. 2 a,b). The mean accumulation rate being there in the period under investigation was approximately 0.017 μmol dm^{-3} yr^{-1} for phosphate and 0.21 μmol dm^{-3} yr^{-1} for nitrate. The increase in the deep water of this station (Fig. 2 c,d) was much lower. The accumulation rates of phosphate and nitrate in winter are roughly half as high with respect to total phosphorus and inorganic nitrogen compounds at the Kattegat stations studied above.

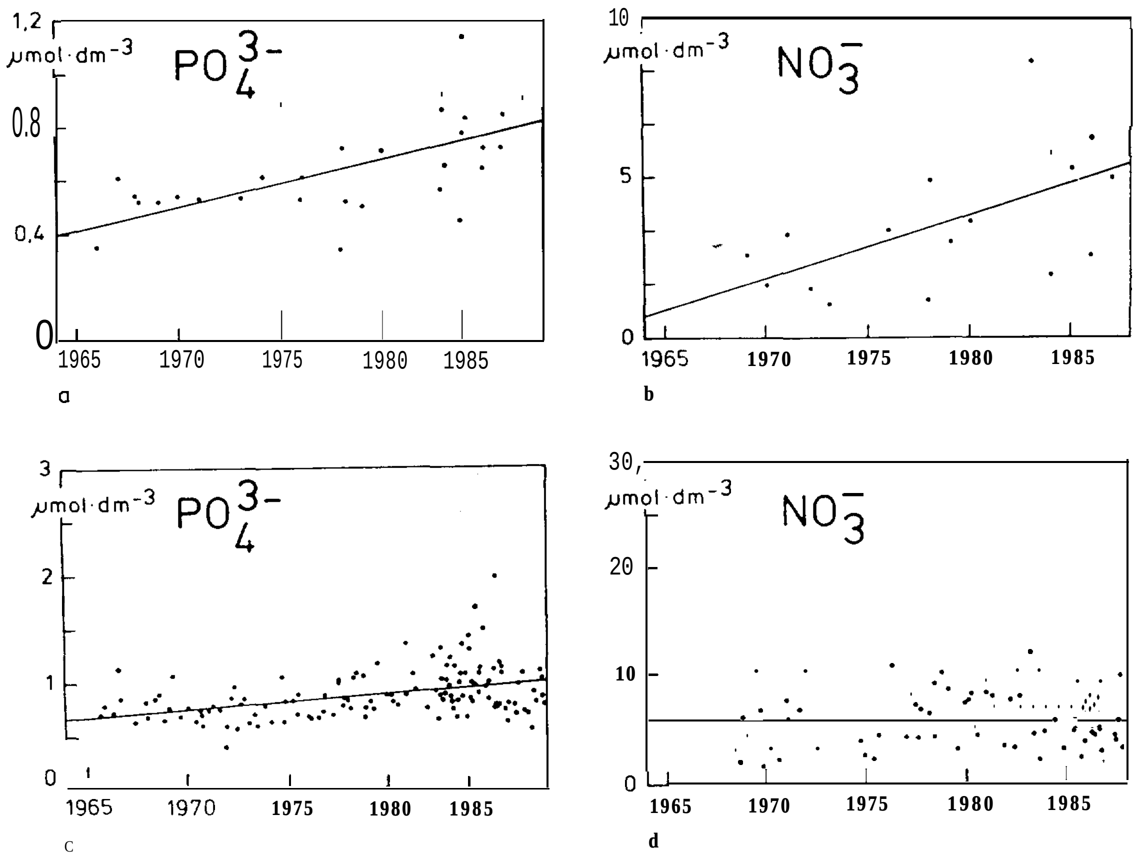

Figure 2. Phosphate and nitrate trends from daily mean values in the surface layer (0-5 m) in winter (a,b) and in the deep water (60-bottom) during the whole year (c,d) at the Kattegat Station Fladen, BMP R6 (by Engström and Fonselius, 1989).

3.3.2 The Belt Sea

H.P. **Hansen**[2], **L.A. Jørgensen**[3], D. **Körner**[4] and D. **Nehring**[1]

The Belt Sea includes the Danish Straits, the Kiel Bay and the Bay of Mecklenburg. One data set used for the following trend analysis in the Belt Sea comes from the Danish National Environmental Research Institute. The investigations were performed under the same suppositions as described for the Kattegat.

The results summarised in Tables 3 and 4 differ not so much from those of the Kattegat. Accumulation rates of the inorganic nitrogen compounds and silicate in winter are very similar in both areas. Positive trends of the nitrogen compounds studied have also been observed in the Belt Sea since about 1982. Silicate concentrations are decreasing in the period under investigation. In contrast to the Kattegat the accumulation rates of total phosphorus strongly variate at the different station5 in the Belt Sea. For this reason the rates of total phosphorus are hardly comparable with the results of the First Periodic Assessment (Baltic Marine Environment Protection Commission, 1987a). No clear changes in the accumulation rates of the inorganic nitrogen compounds have been found in this respect.

The data compiled to assess the Kiel Bay include BMP data as well as national data from Deutsches Hydrographisches Institut (Hamburg), Institut fiir Meereskunde (Kiel) and Landesamt fiir Wasserhaushalt und **Küsten** (Kiel) for the BMP stations **N1** (952 Fehmarnbelt), N3 (Kieler Bucht) and N4 (450) and the national station **"Boknis Eck".** The four stations show corresponding trends in nutrient5 with only minor differences in magnitude. This has been shown already for the period 1957 to 1975 by **Babenerd** (1980). Thus, the trends for the Kiel Bay have been computed on the basis of the combined data of four stations. Winter conditions studied in this area are restricted to January and February. The mean nutrient concentrations in the surface layer between 0 and 10 m depth were used for the investigations.

Figure 3 shows the development of phosphate and inorganic nitrogen (NO_3 + NO_2 + NH_4) concentrations. Some of the nitrogen values before 1975 had to be calculated from reported nitrate concentrations and means of ammonium and nitrite as determined for corresponding months of other years. The calculated trends for the last decade are weak and insignificant. The general feature for the period 1960 to 1989 is similar to other areas of the Baltic Sea. Starting with low values in the 1960 to 1970 decade, we find a drastic increase in concentrations between 1970 and 1980 followed by a period of relatively high levels but insignificant trends. Total phosphorus and silicate concentrations show weak negative trends for the 1980 to 1989 period (Fig. 3b). The mean annual accumulation rate5 calculated for the overall trends are 0.02 μmol dm^{-3} for phosphate and 0.26 μmol dm^{-3} for inorganic nitrogen for the period 1960 to 1989, and -0,07 μmol dm^{-3} for total phosphorus and -0.24 μmol dm^{-3} for silicate for the period 1979 to 1989.

Data from the seasonal cruises of the Institute of Marine Research in Restock-Warnemiinde were analyzed with respect to nutrient trends in the Bay of Mecklenburg using the mean winter concentration5 in the homogeneous surface layer. Figure 4 shows the long-term variations of phosphate and nitrate covering the period 1971-1989. Both nutrient5 are characterized by positive trends. Accumulation rates of 0.014 μmol dm^{-3} yr^{-1} for phosphate and 0.13 μmol dm^{-3} yr^{-1} for nitrate were calculated with respect to the overall trends. These rates are lower as in earlier studies (Nehring and Francke, 1985) depending on the insignificant phosphate changes and the negative nitrate trend (-0.26 μmol dm^{-3} yr^{-1}) in the period 1980-1989. No phosphate and nitrate trends could be identified in the deep water of the Bay of Mecklenburg in the period under investigation.

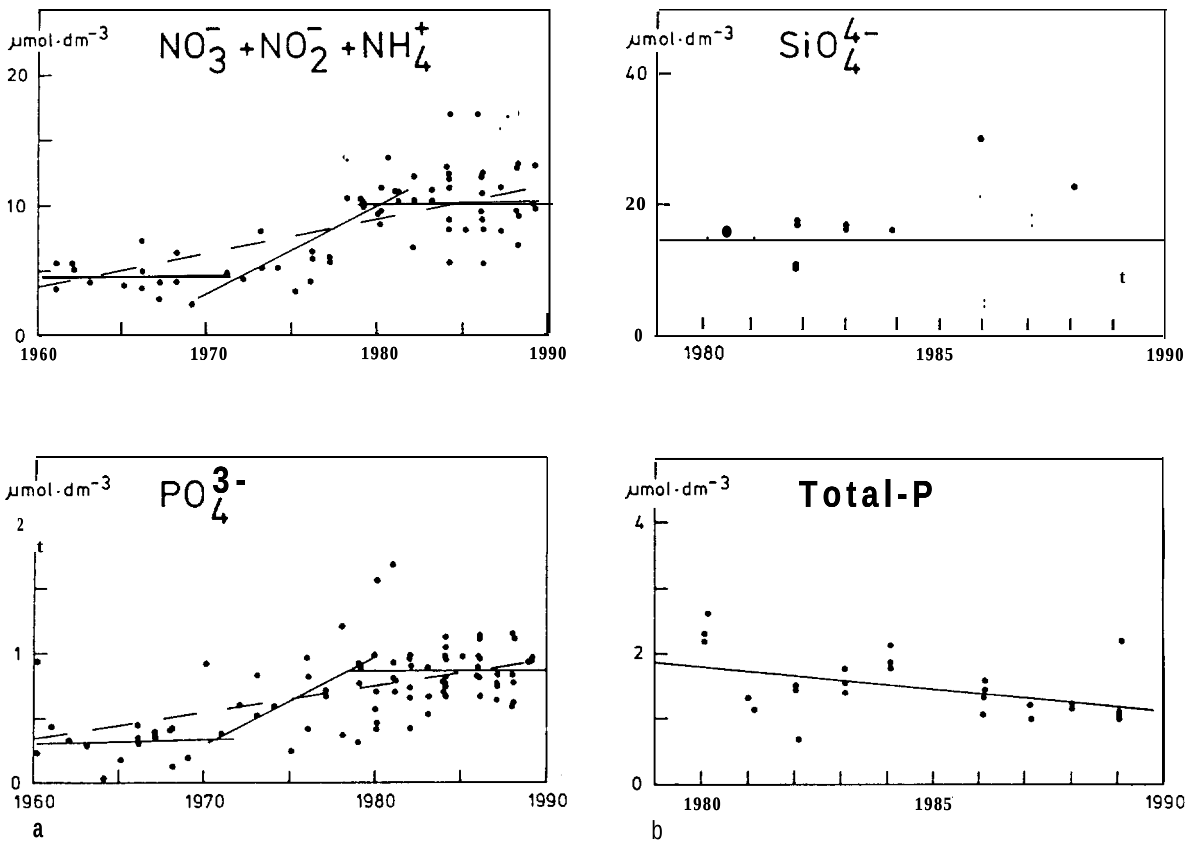

Figure 3. Nutrient trends in the Kiel Bay (Jan - Feb, O-10 m depth), 4 stations pooled

a. inorganic nitrogen and phosphate concentrations from 1960 to 1989,

b. silicate and total phosphorus concentrations from 1979 to 1989.

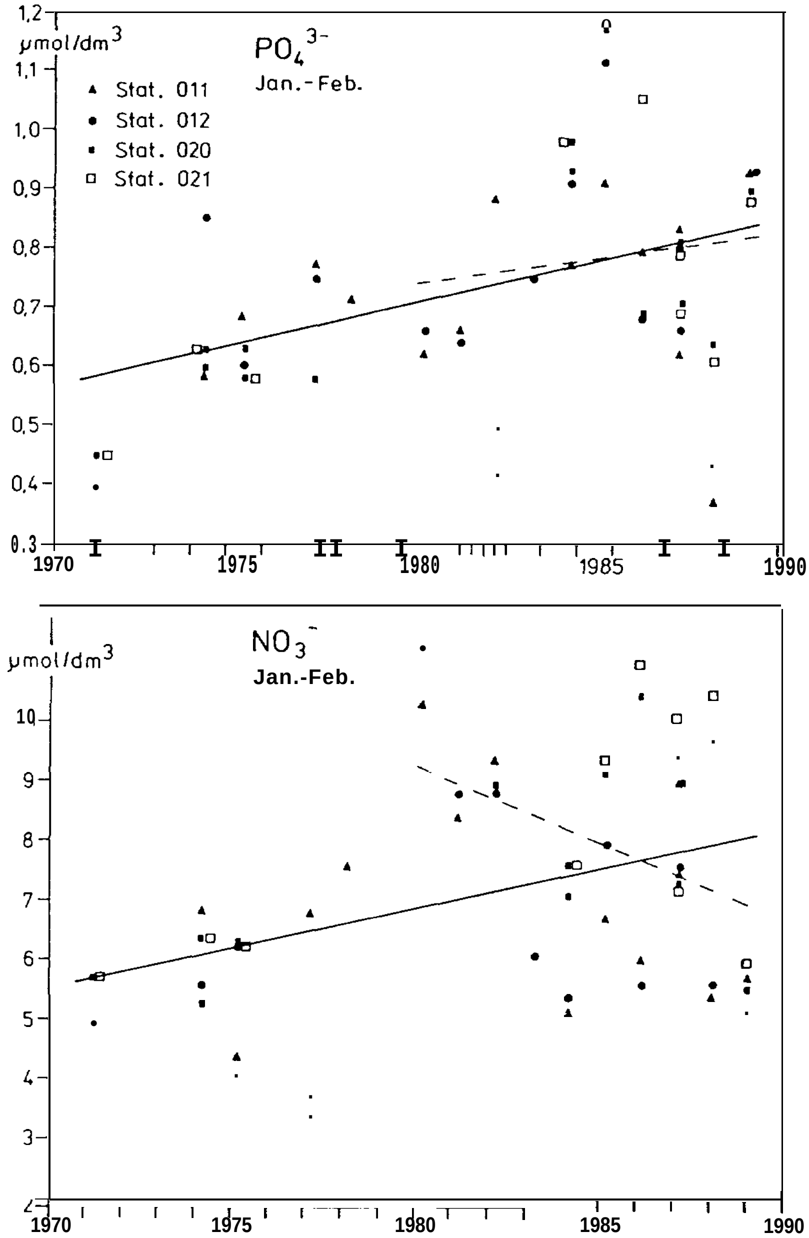

Figure 4. Phosphate and nitrate trends in the winter surface layer of the Bay of Mecklenburg, station 012 = BMP M2 (by Nehring and Matthäus, 1990).

Table 3. Mean annual accumulation rates (trend coefficients) of the inorganic nitrogen compounds (NO_3 + NO2 + NH_4) in the Belt Sea from 1974 - 1988.

BMP Stations	Surface layer (O-10 m) winter data only $\mu mol\ dm^{-3}\ yr^{-1}$	Deepwater (20 m - bottom) year-round data $\mu mol\ dm^{-3}\ yr^{-1}$
P 1 (Halsskov Rev)	(0.45)	(0.11)
Q 2 (Ven)	(0.42)	(0.26)
N 4 (Kjels Nor)	(0.50)	0.13
N 1 (Fehmarn Belt)	0.45	0.15
M 1 (Gedser Rev)	0.25	0.08
Mean	0.41	0.17

Table 4. Mean annual accumulation rates (trend coefficients) of total phosphorus, inorganic nitrogen compounds (NO_3 + NO2 + NH_4), and silicate in the Belt Sea in winter (whole water column).

BMP Stations	tot. P $\mu mol\ dm^{-3}\ yr^{-1}$ 1974-1988	NO_3+NO_2+NH_4 $\mu mol\ dm^{-3}\ yr^{-1}$ 1974-1988	1983-1988	SiO_4 $\mu mol\ dm^{-3}\ yr^{-1}$ 1974-1988
P 1 (Halsskov Rev)	(0.38)	(0.41)	0.51	(-0.38)
Q 2 (Ven)	(0.03)	(0.45)	0.08	(-0.69)
N 4 (Kjels Nor)	(0.07)	(0.47)	0.04	(-0.41)
N 1 (Fehmarn Belt)	(0.01)	(0.45)	0.17	-0.57
M 1 (Gedser Rev)	(0.27)	(0.26)	0.54	(-0.27)
Mean	(0.26)	0.41	(0.31)	-0.46

3.3.3 The Baltic Proper

D. Nehring[1]

The Baltic Proper comprises the Arkona Sea, the Bornholm Sea and the Gotland Sea. Although the Gdańsk Basin and the Western Gotland Sea belong to the last one, they were separately studied due to the responsibility of different scientists (cf. sub-chapters 3.34 and 3.35).

Most of the data used in the following trend studies were collected during monitoring cruises, undertaken by research vessels of the Institute of Marine Research, Restock-Warnemiinde, which have covered the Baltic Proper since 1969. Results from other routine cruises published by Swedish and Soviet institutions and of the International Baltic Year 1969/1970 represented additional sources of data (cf. Nehring and Matthäus, 1990). These permit the study of changes in the phosphate concentrations back as far as 1958.

Sub-trends were calculated in addition to overall trends, mainly to differentiate between the periods preceding and following the last major Baltic inflows during the winters 1975/1976 and 1976/1977. The phase lag of water renewal which corresponds to the distance from the entrance to the Baltic Proper (Darss Sill) and exceed one year, was taken into account by Nehring and Matthäus (1990), when calculating the sub-trends for the different basins.

The amount of data characterizing winter conditions varies in the Baltic Proper. Measurements made between mid-January and mid-April can be used for trend analysis in the south-eastern part of the Gotland Sea. In the Arkona and Bornholm Seas it was decided in each case whether the nutrient concentrations measured in March/April were still representative of the winter situation or had already decreased owing to the spring development of the phytoplankton. The period under investigation includes also the winter conditions in 1989.

Regional differences in winter concentrations in the surface layer of the offshore Baltic regions are generally slight. As in earlier studies (Nehring, 1985), the values measured at all stations in a given sea area could therefore be treated as a uniform data set (cf. Wulff and Rahm, 1989). Regional differentiation was nevertheless undertaken between the Arkona and Bornholm Seas and the south-eastern Gotland Sea.

The mean nutrient concentrations were calculated for the homogeneously mixed surface layer in winter and used in the following trend investigations. The thickness of the mixed winter surface layer varied between 10-30 m in the more shallow Arkona Sea and 50-60 m in the basins of the Central Baltic. Its extend is also affected by actual hydrographic conditions. Between 1958 and 1966 only surface values were published and used in the trend calculations.

Figure 5 shows characteristic sub-trends and the overall trends for phosphate and nitrate concentrations in the surface layer of the south-eastern Gotland Sea. Similar figures, but with data more scattering around the regression lines, were published by Nehring and Matthäus (1990) for the Arkona and Bornholm Seas.

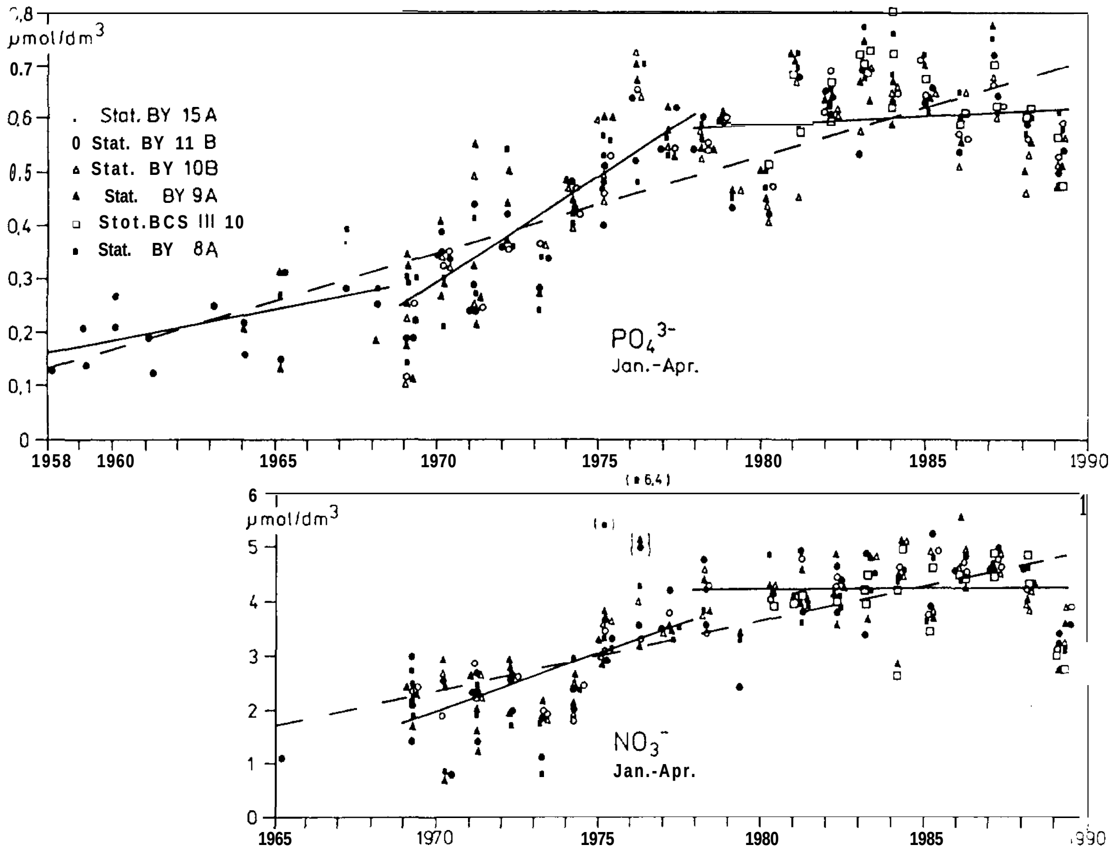

Figure 5. Nutrient trends in the surface layer of the south-eastern **Gotland** Sea in winter, Station BY 15 = BMP **J1;** Station BCS III 10 = BMP **K1** (Nehring and **Matthäus,** 1990).

Positive overall phosphate and nitrate trends which are significant by t-test were identified in the surface layer of all regions under investigation. Regional differences of the overall trends and the sub-trends for the different regions were slight in most cases, as shown by the mean annual accumulation rates in Table 5. The rates of the nutrient increase in the separate periods, ***however,*** was characterized by great differences. The rapid increase from 1969 to 1978 in all areas under investigation was followed by a period in which the accumulation rates of phosphate and nitrate were much lower. In many cases the changes ***were*** no longer significant and winter concentrations are marked by considerable short-term variations (Fig. 5). In the Arkona Sea, the nitrate concentration even decreased slightly, on average, between 1978 and 1989 (Table 5).

Table 5. Mean annual accumulation rates (trend coefficients) of phosphate and nitrate in the surface layer of the Baltic Proper in winter.

Regions	Periods	PO_4 $\mu mol\ dm^{-3}\ yr^{-1}$	NO_3 $\mu mol\ dm^{-3}\ yr^{-1}$
Central Arkona Sea	1969-1978	0.043	0.21
(Stat.BMP K7=BY1,	1978-1989	(0.006)	(-0.02)
BMP K4=BY2,102,	1965-1989		0.13
103,110,111	1964-1989	0.021	
BMP K5=113)			
Central Bornholm Sea	1969-1978	0.046	0.15
(Stat.BY4,BMP K4=BY5,	1978-1989	(0.003)	(0.04)
BY6, 214)	1965-1989		0.13
	1958-1989	0.021	
Southeastern	1958-1968	0.012	
Gotland Sea	1969-1978	0.041	0.23
(Stat.BY8,BMP K1=	1978-1989	(0.004)	(0.01)
BCS III 10,BY9,BY10,	1965-1989		0.13
BY11,BMP J1=BY15)	1958-1989	0.018	

The positive overall trends in the winter surface layer covering the periods 1958-1989 for phosphate and 1968-1989 for nitrate therefore result mainly from the considerable nutrient increase between 1969 and 1978. The trends in the Arkona Sea, the Bornholm Sea, and the Eastern Gotland Sea are interrupted by decreasing phosphate and nitrate concentrations from 1976 to 1980 and 1984 to 1989.

The deep water of the Arkona Basin is seasonally renewed. No phosphate and nitrate trends have been discovered in this layer.

Stagnation prevails in the deep water of the Central Baltic Proper basins beneath the permanent halocline. Conditions may be oxic or anoxic, depending on the depth and on water renewals.

The Bornholm Basin is the most western sub-area of the Baltic Proper containing stagnant deep water which is not annually renewed. The thickness of the stagnant water body is slight in this basin and characterized by the occasional formation of hydrogen sulphide.

When calculating the phosphate and nitrate trends in the deep water of the Bornholm Basin, a distinction was made between values that definitely reflected the influence of anoxia and those that were affected little or not at all by anoxic conditions (cf. Nehring, 1989). For this reason the threshold concentration of 3 μmol dm^{-3} was chosen for phosphate. In addition, the trend was also calculated for all observations. In the case of nitrate, the trend calculation was based only on values above 2 μmol dm^{-3}.

The long-term variations of parameters in the near-bottom water layer of the Bornholm Deep are presented in Figure 6. No sub-trends were identified in the distribution of phosphate or nitrate. Both nutrients exhibit positive overall trends that are significant (Table 6). The phosphate trend is mainly governed by the release from the sediment (PO_4 > 3 μmol dm^{-3}) associated with the more common occurrence of anoxic conditions since the mid-seventies. Phosphate accumulation, however, which can be considered a direct consequence of eutrophication and is characterized by the trend of the phosphate values < 3 μmol dm^{-3}, is much lower.

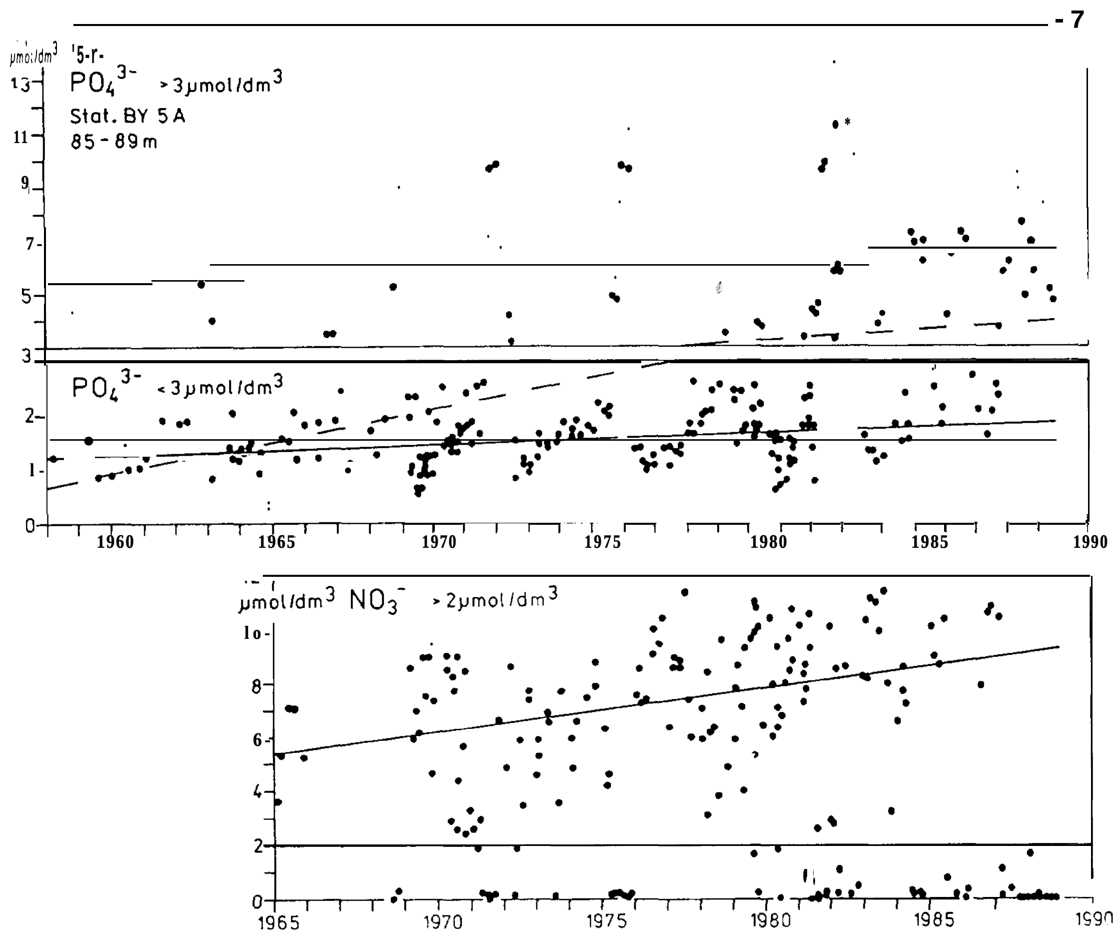

Figure 6. Nutrient trends in the near-bottom water layer of the Bornholm Deep, Stat. BY 5 = BMP K2 (Nehring and Matthäus, 1990).

Anoxic conditions have been recorded rarely, if at all, at a depth of 100 m in the Gotland Sea. Hydrogen sulphide is encountered more frequently, however, as the depth increases. The long-term variations under predominantly oxic and anoxic conditions, respectively, are shown in Figures 7 and 8 for selected depths below the halocline in the Gotland Deep. According to Nehring and Matthäus (1990) similar figures are available for the Fårö Deep.

Two periods with different trends were distinguished not only for nutrients but also for salinity and oxygen (cf. chapters 1 and 2, cf. Nehring and Matthäus, 1990). The different parameters also vary individually in both time and depth. The phosphate and nitrate concentrations increased rapidly, on average, at a depth of 100 m at the beginning of the investigations, whereas the trend during the second period, which started between 1976 and 1978, was less pronounced (Fig. 7) and, according to the t-test, sometimes insignificant (Table 6).

Changes in phosphate concentrations related to the different stagnation periods become increasingly apparent at depth greater than 150 m. They are superimposed on the long-term variations and are encountered particularly in the near-bottom layers where oxic and anoxic conditions alternate. In the Gotland Deep (Fig. 8) a decrease that was insignificant according to the t-test was observed at the beginning of the period studied (Table 6, 200 m). The period from 1977 to 1988, however, was characterized by a major increase in phosphate concentrations. This trend mainly results from the remobilization of phosphate by reduction of the iron hydroxo complex in the sediments caused by the increasing hydrogen sulphide concentrations (cf. Chapter 2) and consequently the decreasing redox potential. Relations to the recent long-lasting stagnation period are obvious. The overall trend in the near-bottom water layer was insignificant at this station.

The mean annual accumulation rates (trend coefficients) of phosphate and nitrate summarized in Tables 5 and 6 show relatively small differences in the separate regions of the Baltic Proper. With respect to the surface layer in winter, this means that the changes in the trophic level, characterized by the nutrient concentrations are quite similar in all areas under investigation. This is also true in the trend behaviour of phosphate and nitrate in the separate deeps, although differences exist in the mean annual variation rates. Exceptions are the western basins of the Baltic Proper. No trend has been discerned in the deep water of the Arkona Basin, because of its seasonal water exchange. The near-bottom water layer in the Bornholm Basin is rather often renewed in comparison with the Eastern Gotland Basin which is characterized by predominating stagnant conditions below 125-150 m depth since the last major inflow in 1976/1977. For this reason, no sub-trends in phosphate and hydrographic parameters (Nehring and Matthäus, 1990) could be identified in the deep water of the Bornholm Basin.

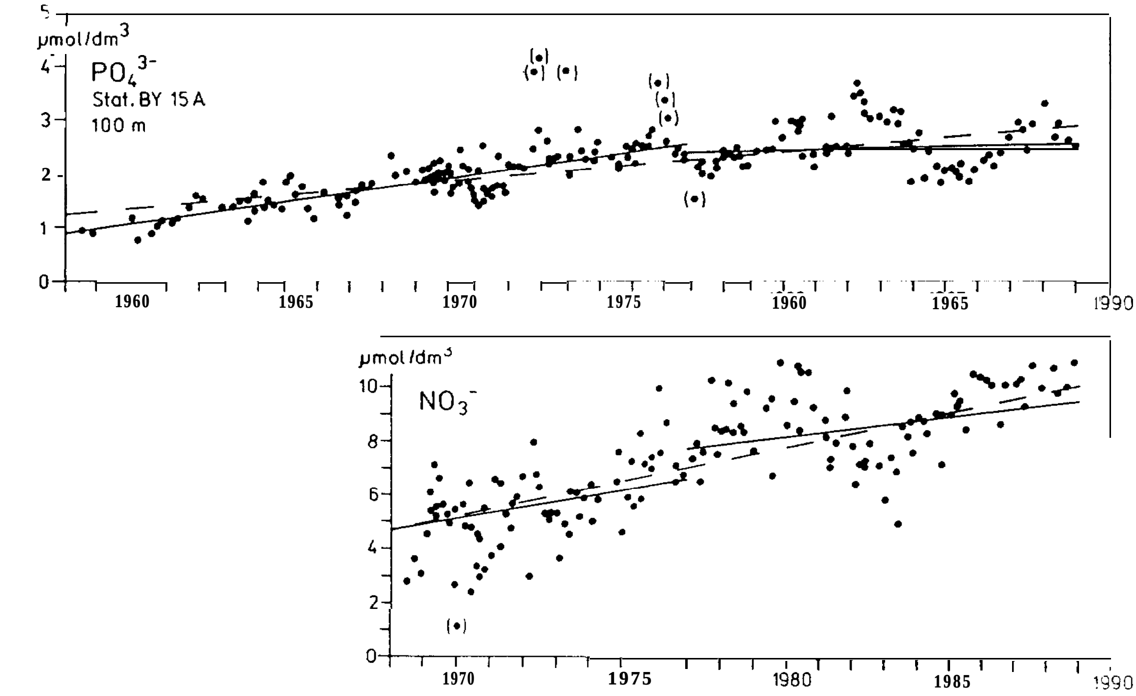

Figure 7. Phosphate and nitrate trends in the intermediate water layer of the Gotland Deep, Stat. BY 15 = BMP J1, from Nehring and Matthäus 1990.

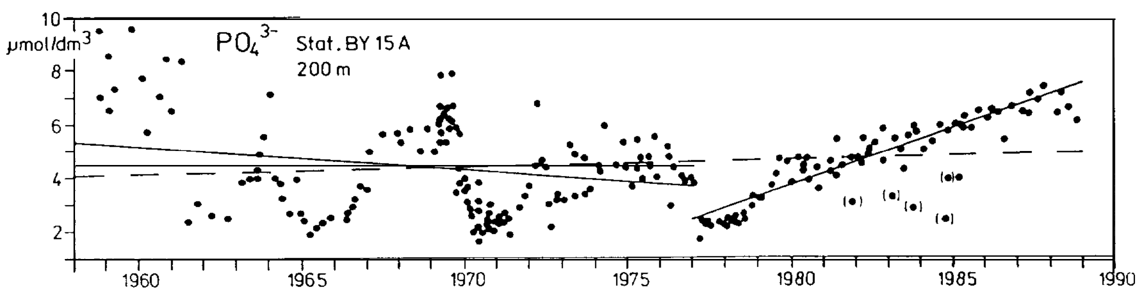

Figure 8. Phosphate trends in the near-bottom water layer of the Gotland Deep, Stat. BY 15 = BMP J1, from Nehring and Matthäus, 1990.

Table 6. Mean annual accumulation rates (trend coefficients) of phosphate and nitrate in central Baltic deep waters.

BMP **Stations**	Depths m	Periods	PC4 µmol dm^{-3} yr^{-1}	NC3 µmol dm^{-3} yr^{-1}
K2	85-89	1958-1978		
(Bornholm Deep,		1976-1988		
BY 5)		1958-1988	0.021 (<3)	
			(0.046, >3)	
			0.120 (3)	
		1965-1988		0.17(<2)
J 1	100	1958-1976	0.086	
(Gotland Deep,		1968-1976		0.21
BY 15)		1977-1988	(0.021)	0.15
		1968-1988		0.26
		1958-1988	0.054	
	200	1958-1976	-0.086	
		1977-1988	0.414	
		1958-1988	(0.028)	

Salinity is continuously decreasing in the Baltic Proper since the last major inflows of highly saline water in the late 1970s (Chapter 1, cf. Nehring and **Matthäus,** 1990). Although the winter concentrations of phosphate and nitrate are no longer increasing, on average, in the surface layer, the close positive correlation between these nutrients and the salinity observed in earlier investigations (Nehring 1985, Baltic Marine Environment Protection Commission, 1987a) do not longer exist in recent times.

Trends of total phosphorus and silicate covering the period between 1974 and 1988 were studied by Jorgensen (1989, Danish National Environmental Research Institute, unpublished) in the Arkona Sea in winter. The results basing on the mean concentrations in the whole water column are summarised in Table 7.

The accumulation rate of total phosphorus is more than twice as high as that of the overall phosphate trend mentioned above (Table 5) for the winter surface layer in this region.

Silicate concentrations are decreasing, on average, in the Arkona Sea. The trend coefficient of this nutrient is roughly half as high in comparison with those in the Kattegat and the Belt Sea.

Table 7. Mean annual accumulation rates (trend coefficients) of total phosphorus and silicate in the Arkona Sea in winter from 1974-1988 (L.A. Jorgensen, unpublished; whole water column).

BMP Stations	tot. P $\mu mol\ dm^{-3}\ yr^{-1}$	SiO_4 $\mu mol\ dm^{-3}\ yr^{-1}$
K 6 (Stevens)	0.041	-0.19
K 7 (BY 1)	0.061	-0.29
Mean	0.051	-0.24

3.3.4 Gdańsk Basin

A. Trzosińska[4]

The Gdafisk Basin was studied on the basis of data collected by the Institute of Meteorology and Water Management in Gdynia, Poland. Trend studies on the nutrient accumulation were carried out at Station BMP L1 (Gdańsk Deep = P1) well representing the deep water area of the Gdańsk Basin. Phosphate concentrations have been measured from 1961 at this station, whereas nitrate and silicate data are available since 1969.

The winter conditions in the surface layer were characterized by measurements from the beginning of January to March, 20th. It has been checked that trend calculations carried out for the layer O-10 m (Table 8) and for the layer O-40 m (Baltic Marine Environment Protection Commission, 1987a) led to the same findings. The results of the trend studies in the winter surface layer are summarized in Figure 9 and Table 8.

Studies on the phosphate accumulation in the winter surface waters yield two sub-trends being positive in the period 1961-1977 and negative in the period 1978-1988. The high accumulation rate in the first period is responsible for the positive overall trend.

Nitrate concentrations are increasing, on average, in the winter surface layer of the Gdańsk Deep from 1971to 1988. The overall trend of silicate concentrations was negative in the period 1969-1988, irrespective of the season. No significant sub-trends could be identified for both nutrients.

Figure 10 and Table 8 show that nitrate and silicate concentrations in the intermediate water layer with permanent oxic conditions (80 m) are characterized by a similar trend behaviour as in the winter surface layer. Whereas the sub-trends of phosphate also agree in both layers, the overall trend is negative thus deviating in this case from the surface layer.

Table 8. Mean annual accumulation rates (trend coefficients) of nutrients ($\mu mol \cdot dm^{-3}\ yr^{-1}$) in the Gdańsk Deep (Stat. BMP L1 = P1; only winter concentrations were used in the layer 0-10 m depth, except for silicate during 1969-1988 and total phosphorus).

Periods	Depths/m	PO_4	P_{tot}	NO_3	SiO_4	Remarks
1961-1977	0-10	0.027				
1978-1988	0-10	-0.016				
1961-1988	0-10	0.007				
1979-1984	0-10		(0.031)			
1985-1988	0-10		(-0.003)			
1979-1988	0-10		-0.017			
1971-1988	0-10			0.15		
1974-1988	0-10				-0.54	
1969-1988	0-10				-0.42	
1961-1976	80	0.123				
1977-1988	80	-0.058				
1961-1988	80	-0.018				
1979-1988	80		-0.124			
1969-1988	80			0.15	-1.58	
1961-1976	100-108	0.342				
1977-1988	100-108	-0.144			(0.48)	
1961-1988	100-108	(0.032)				
1979-1988	100-108		(-0.112)			
1980-1988	100-108			(-0.29)		
1969-1988	100-108			(0.06)	-1.24	
1961-1976	100-108	0.134				$O_2 \geq 0$
1977-1988	100-108	(-0.055)				$O_2 \geq 0$
1961-1988	100-108	0.028				$O_2 \geq 0$
1969-1979	100-108			0.32		$O_2 \geq 0$
1980-1988	100-108			0.43		$O_2 \geq 0$
1969-1988	100-108			0.28	-1.22	$O_2 \geq 0$
1977-1988	100-108				1.43	$O_2 \geq 0$

Oxic and anoxic conditions alternate in the near-bottom water layer (100-108 m) of the Gdańsk Deep. The results of the investigations in this layer are shown in Figure 11 and Table 8. In the presence of oxygen, the sub-trends of phosphate were positive in the period 1961-1976 and negative in the period 1977-1988, generating a positive overall trend. The results are similar, but insignificant by t-test for the overall trend, when phosphate concentrations measured under oxic and anoxic conditions were used.

A strong positive overall trend in nitrate concentrations was identified in the near-bottom water layer when excluding measurements performed under anoxic conditions. Sub-trends do not differ very much from that trend. In agreement with the trends in the other layers, silicate concentrations are also generally decreasing in the bottom water. Significant accumulation of silicate can be found there for the present stagnation period, provided that only oxic conditions have been taken into account (Table 8).

Concentrations of total phosphorus studied in the Gdaiisk Deep since 1979 generally follow the trend behaviour of phosphate in the different water layers during the period under investigation (Figs. 9-11, Table 8).

The degree to which the surface waters of the Bay of Gdaiisk are affected by the river run-off depends on river outflow and water circulation imposed by the meteorological conditions (Cyberska and Krzymiński, 1988). It happens, however, very rarely that Vistula River water can be traced in the Gdaiisk Deep area.

Whereas the close positive correlation between phosphate concentrations and salinity continues recently in the winter surface layer of the Gdańsk Deep (cf. HELCOM, 1987a), nitrate differs in this respect, because in the period 1978-1988 the declining salinity (-0.046 o/oo annually) was accompanied by increasing nitrate concentrations.

The mean 5-year amplitude between the winter maximum and the spring minimum in silicate concentrations amounts to 12-18 μmol dm^{-3} depending on the distance from the Vistula River mouth (Trzosiiiska, 1990). the extremely high concentrations of this nutrient measured in the area of the Gdaiisk Deep in the beginning of 1974 and 1975 as well as in early spring 1988 (Fig. 9) were probably connected with the river run-off.

The decreasing accumulation rate of nitrate and the negative overall-trend marked in the phosphate, total phosphorus and silicate concentrations in the intermediate water layer, identified recently not only in the Gdaiisk Deep but also in the Central Baltic basins (cf. sub-chapter 3.33) are probably caused by intensified vertical mixing, which is also reflected by the improvement of the oxygen conditions (cf. Chapter 2). Variations caused by these processes in the nutrient distribution were even more pronounced in the near-bottom water layer with alternating oxic and anoxic conditions. They are also favoured by the decreasing salinity (cf. Chapter 1) facilitating the advective water exchange in the Gdaiisk Deep.

The elimination of the data measured in the presence of hydrogen sulphide confirmed the direction of the phosphate trends but lowered considerably the mean annual accumulation rates. The overall trend of phosphate identified in the bottom water of the Gdańsk Deep during oxic conditions was well comparable with those for the Bornholm and Gotland Deeps (Nehring, 1989), whereas the nitrate trend was more pronounced in comparison with the respective trend in the Bornholm Deep.

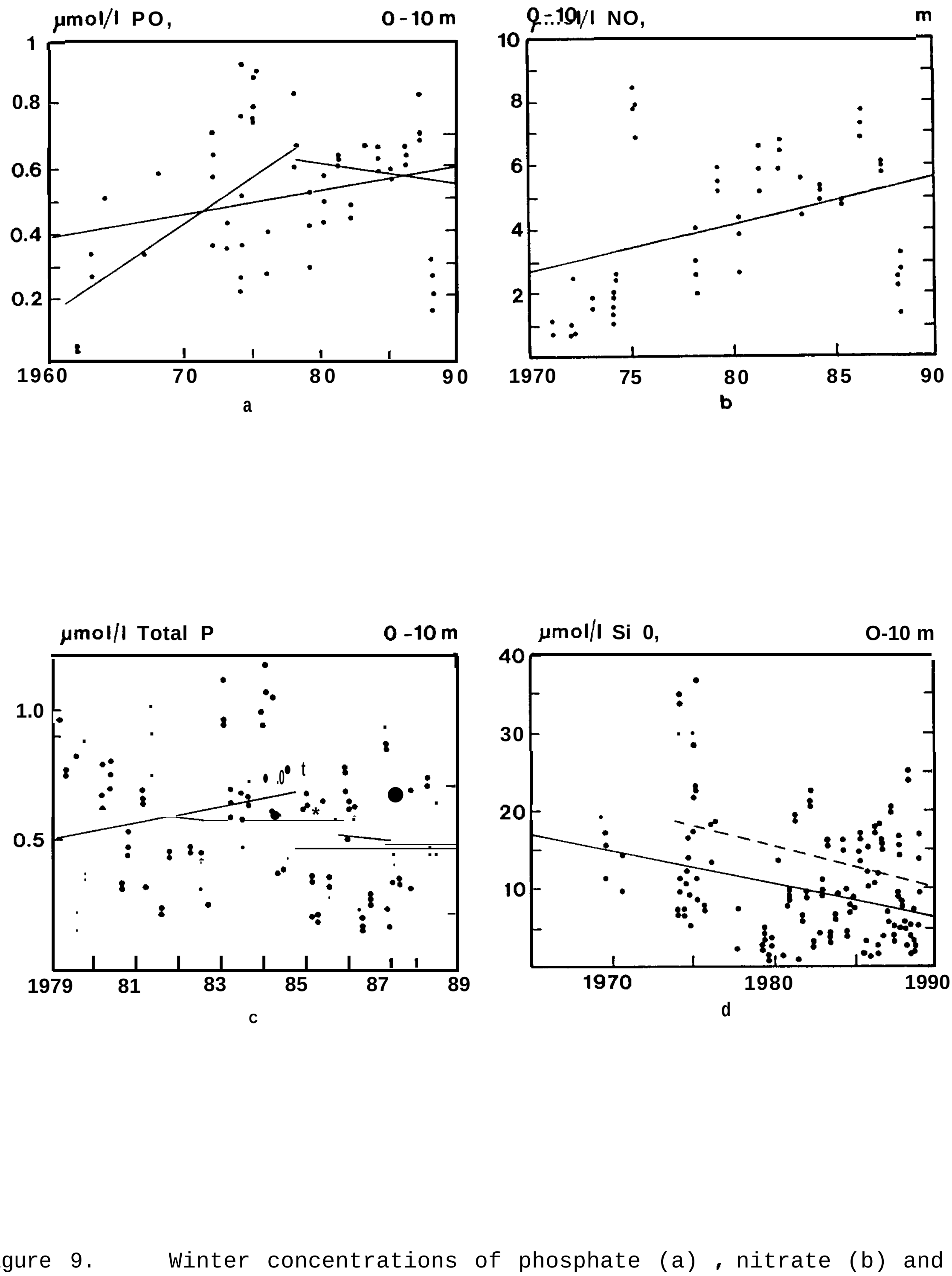

Figure 9. Winter concentrations of phosphate (a) , nitrate (b) and silicate (d- broken line) and year-round concentrations of total phosphorus (c) and silicate 1969-1988 (d) in the surface layer (O-10 m depth) of the Gdaiisk Deep (Stat. BMP L1 = Pl).

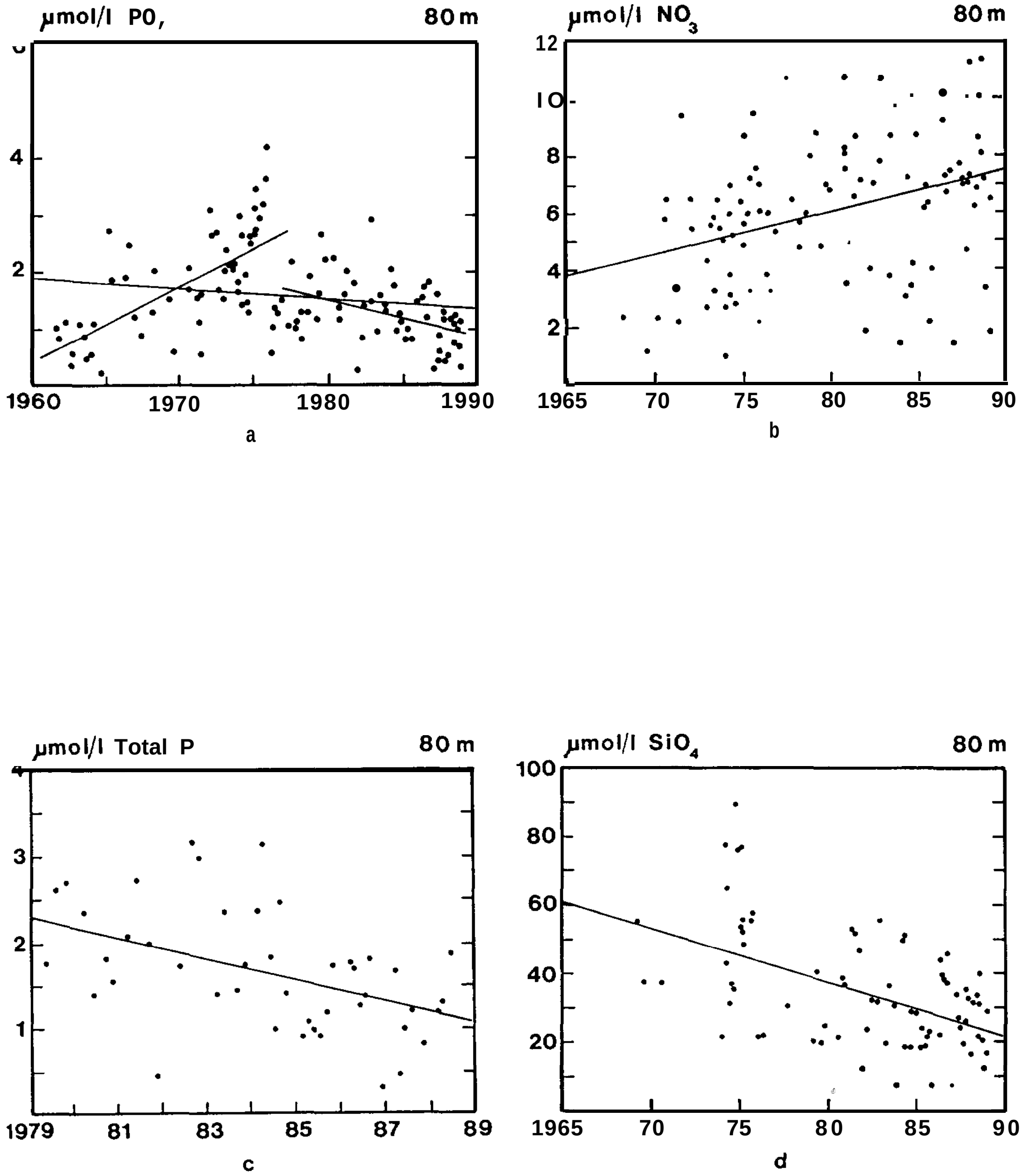

Figure 10. Nutrient concentrations in the intermediate water layer (80 m depth) of the Gdańsk Deep (Stat. BMP L1 = P1).

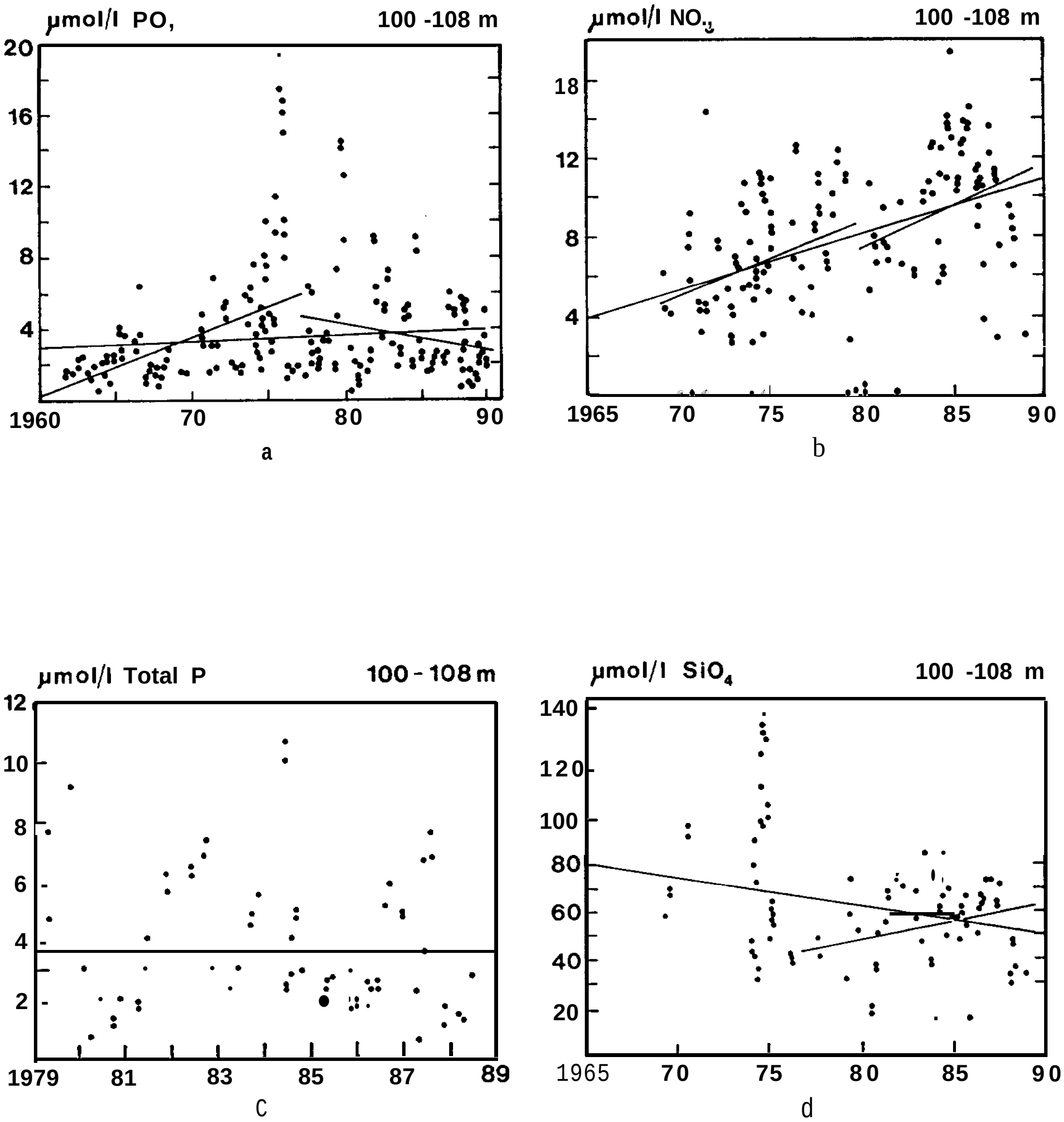

Figure 11. Nutrient concentrations in the near-bottom water layer (100–108 m depth) of the Gdańsk Deep (Stat. BMP L1 = P1; only concentrations measured under oxic conditions were used in the case of nitrate and silicate).

3.3.5 Western Gotland Sea
F. Wulff[8]

The Western Gotland Sea includes the BMP stations Landsort Deep (H 3 = BY 31) and Karlsö Deep (I 1 = BY 38). The surface distribution of both inorganic and total nitrogen as well as total phosphorus studied in winter show increasing concentrations for the entire period at the Landsort Deep whereas the silicate concentrations are decreasing (Fig. 12a). The same trends can be seen below the halocline at 100 m (Fig. 12b) and at 400 m (Fig. 12c) but are modified by the changing hydrographic and oxygen conditions. Here, nitrate maxima and phosphate minima are found in September 1978 and October 1985, after the intrusion of oxygenated water masses (cf. Chapters 1 and 2).

The same trend behaviour but less clear as that in the Landsort Deep was found in the surface layer of the Karlsö Deep in winter. None of the stations in the Western Gotland Sea show any decreasing trends in phosphate and nitrate concentrations. At 100 m depth, no clear trend in the phosphorus fractions can be seen whereas both inorganic and total nitrogen are increasing and silicate is decreasing but with a considerable scatter due to variable oxygen conditions in the deep waters.

3.3.6 Gulf of Bothnia
F. Wulff[8] and M. Perttilä[6]

The Gulf of Bothnia includes the Åland Sea, the Bothnian Sea, and the Bothnian Bay. In the Åland Sea (BMP D1 = F 64) the short sampling period and limited frequency make it difficult to discern any clear trends in wintertime surface concentrations (Fig. 13a). However, at 200 m depth where data for a longer period are available (1959-1988) there are positive trends in both phosphate and total phosphorus or nitrate and inorganic nitrogen concentrations (Fig. 13b). The trend for (decreasing) silicate concentrations is less pronounced. The same conclusions hold for the surface water observations from the Bothnian Sea (BMP C4 = SR 5) as well (Fig. 14a). At 100 m depth (Fig. 14b), the nitrate concentrations have increased about 3 times and the phosphate concentrations have doubled in this area during the last 20 years.

Oceanographic operations are difficult and expensive in ice-covered areas. This is at least one reason for the restricted pool of nutrient data measured in the northernmost parts of the Baltic Sea in winter.

The observations from the Bothnian Bay (BMP Al = F 2) in the HELCOM data base are too few to discern clear long-term wintertime trends. However, the nitrate trend including data from all seasons is very clear at 30 m depth (cf. Fig. 17). Wulff and Rahm (1988) showed that silicate concentrations have decreased and that total amounts of nitrogen have increased, particularly during summer if the total mean amounts for 1971-1981 are compared. No clear trends for phosphorus fractions are obvious in this analysis.

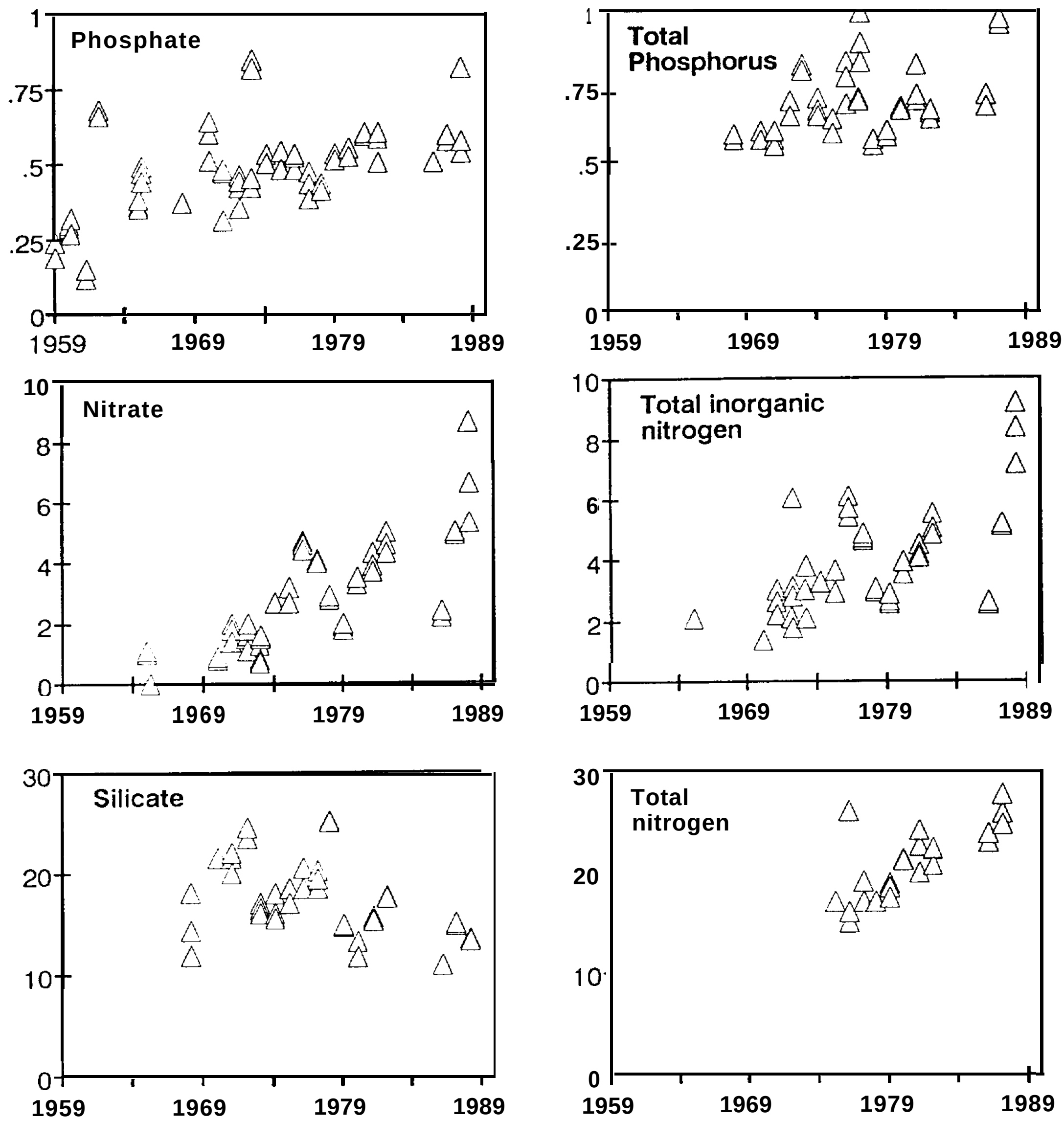

Figure 12a. Long-term variations of nutrient concentrations at the Station Landsort Deep (BMP H3 = BY 31); winter concentrations (15 Jan - 15 Apr) in the surface layer (0-12 m).

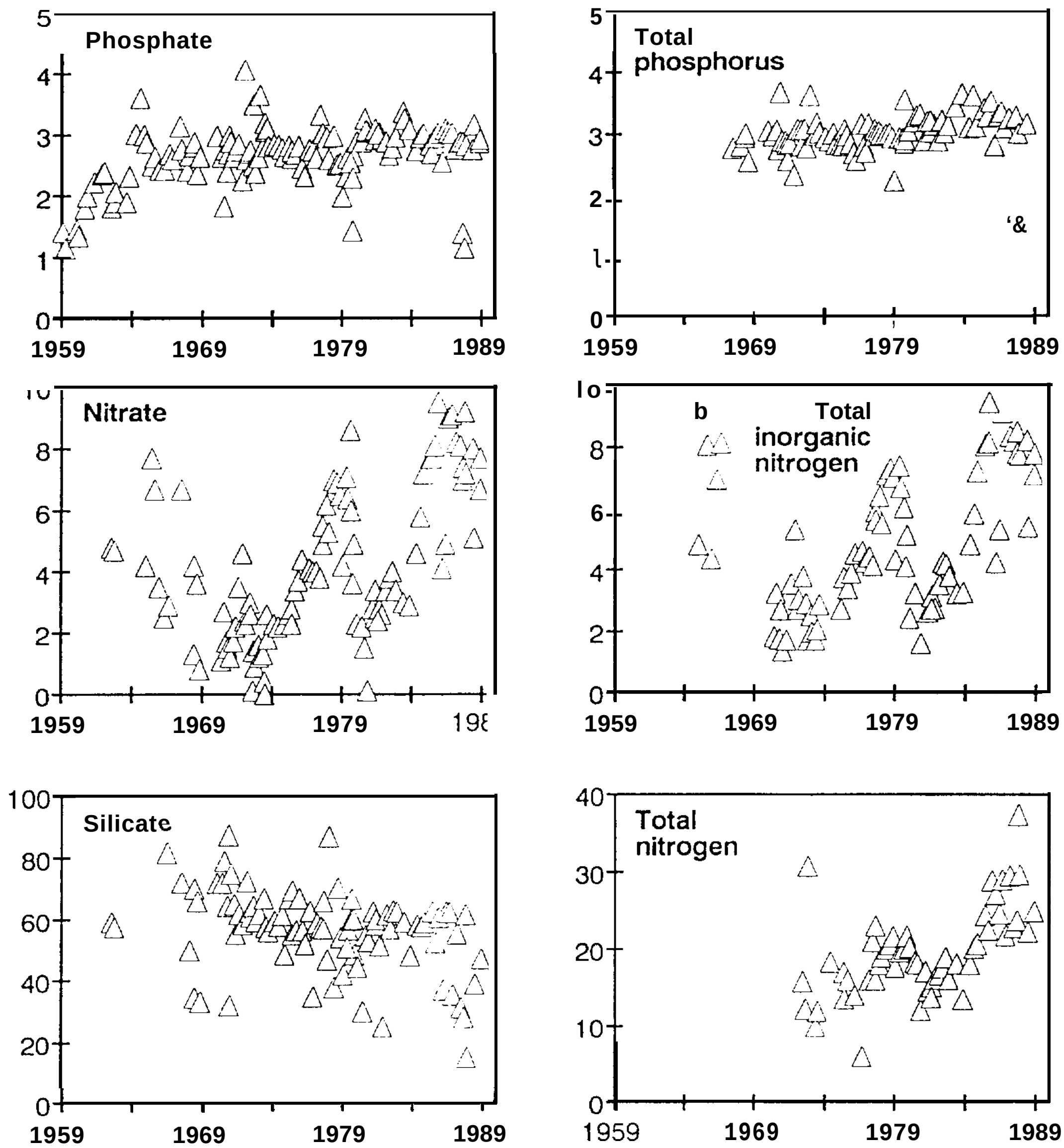

Figure 12b. **Long-term variations of nutrient concentrations at the Station Landsort Deep (BMP H3 = BY 31) at 100 m depth (year-round data).**

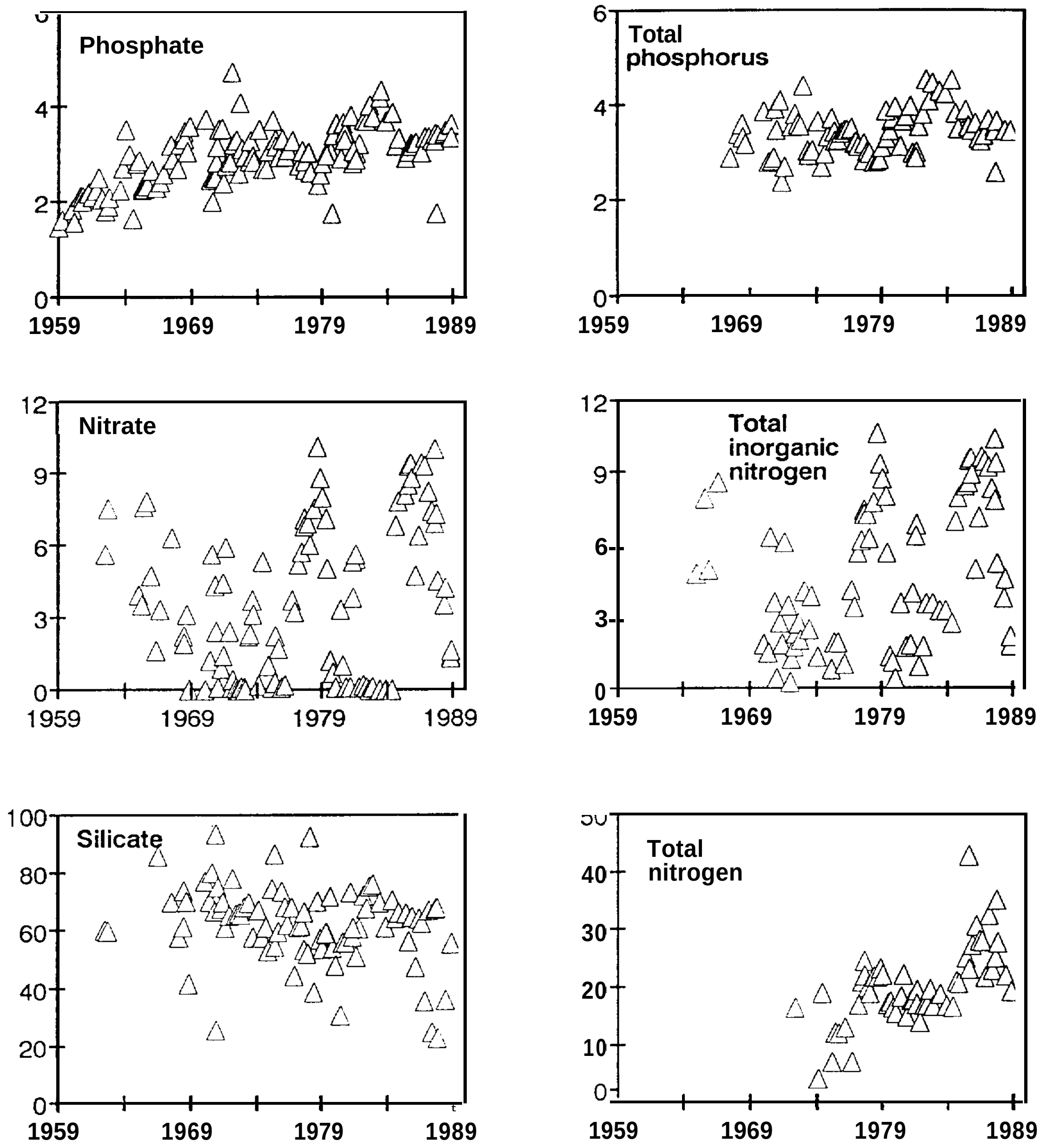

Figure 12c. Long-term variations of nutrient concentrations at the Station Landsort Deep (BMP H3 = BY 31) at 400 m depth (year-round data).

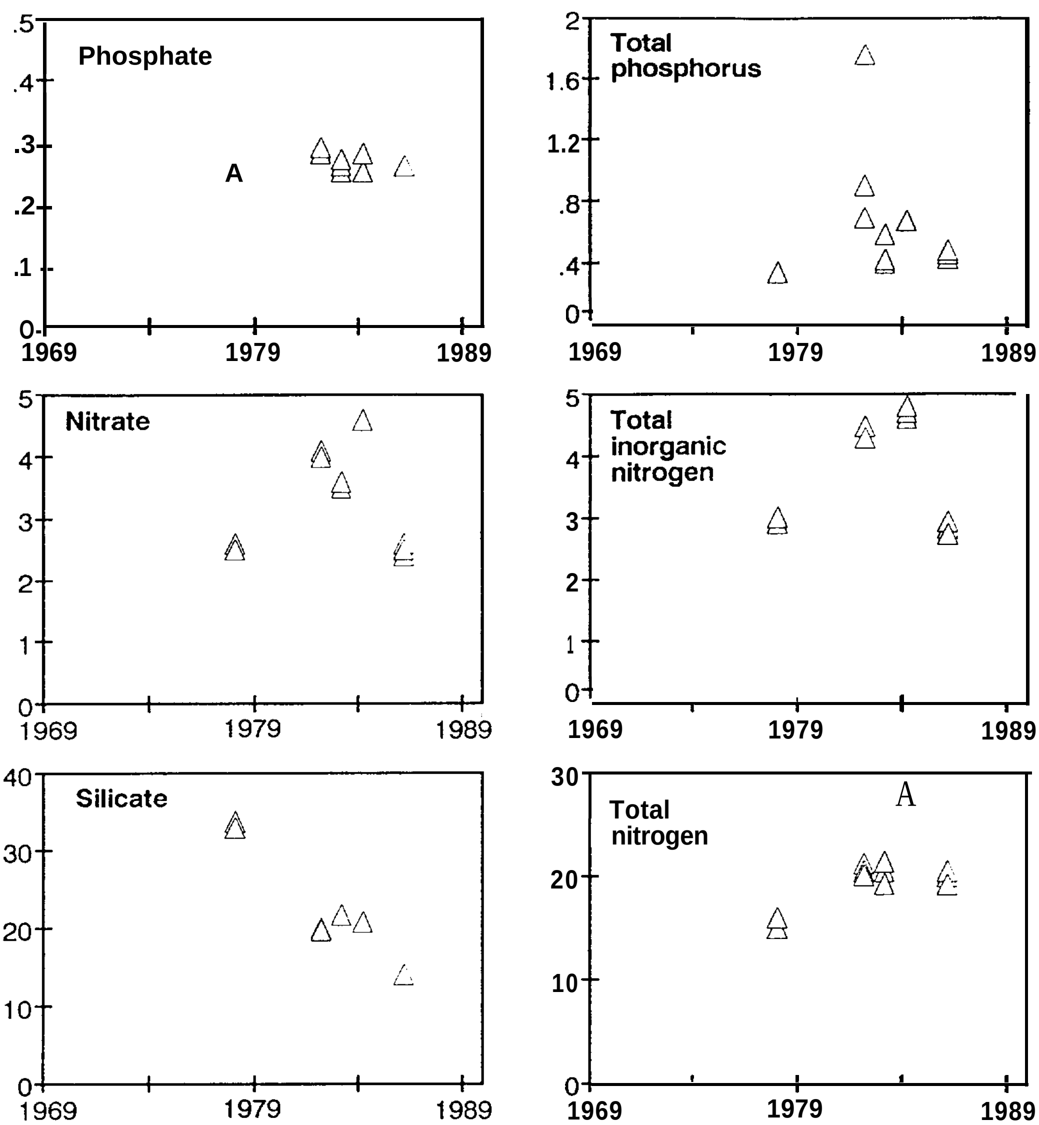

Figure 13a. Long-term variations of nutrient concentrations in the **Åland** Sea **(BMP D1 = F 64);** winter concentrations (1 Jan – 31 Mar) in the surface layer (0-12 m).

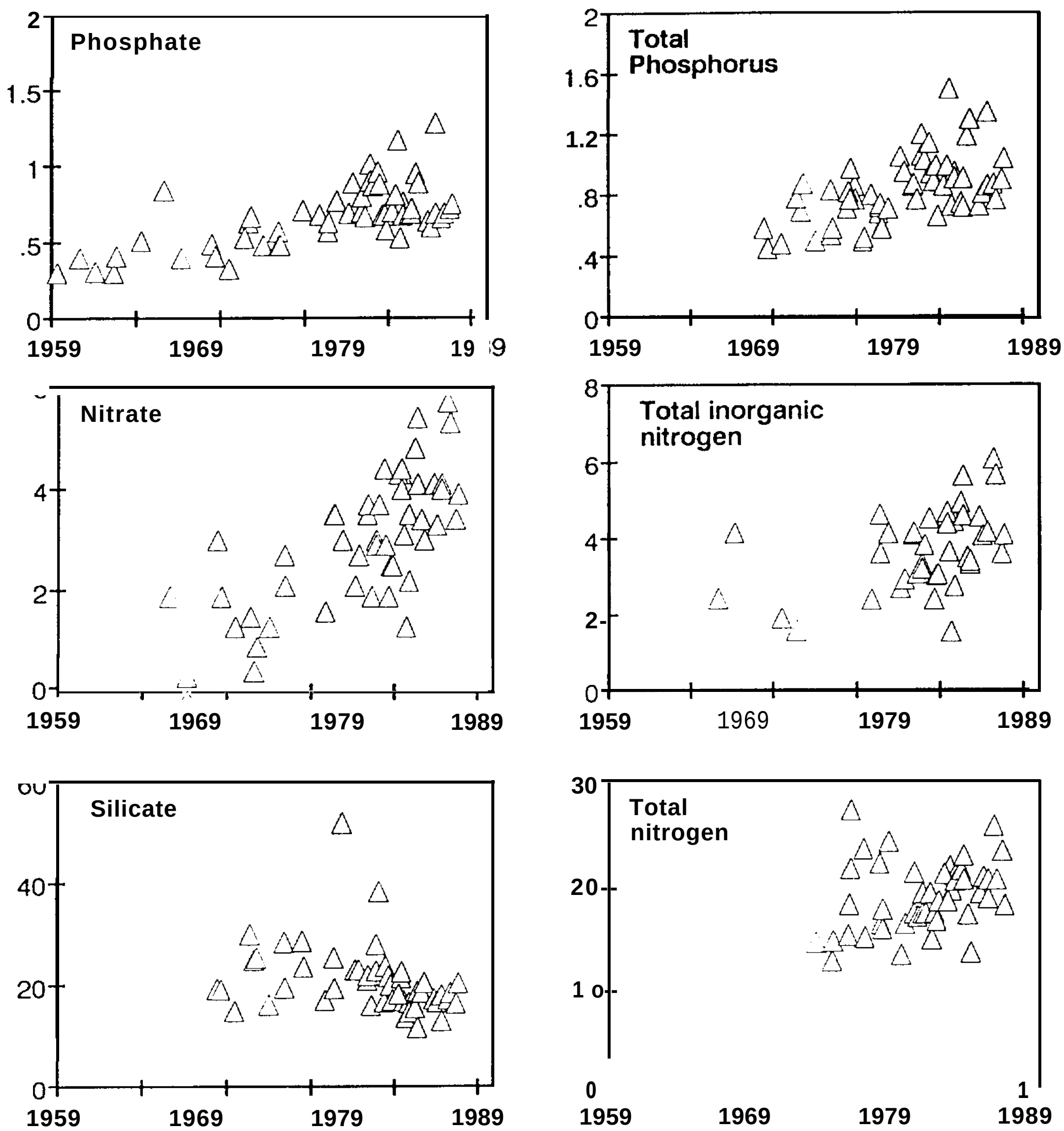

Figure 13b. Long-term variations of nutrient concentrations in the Åland Sea (BMP D1 = F 64) at 200 m depth (year-round data).

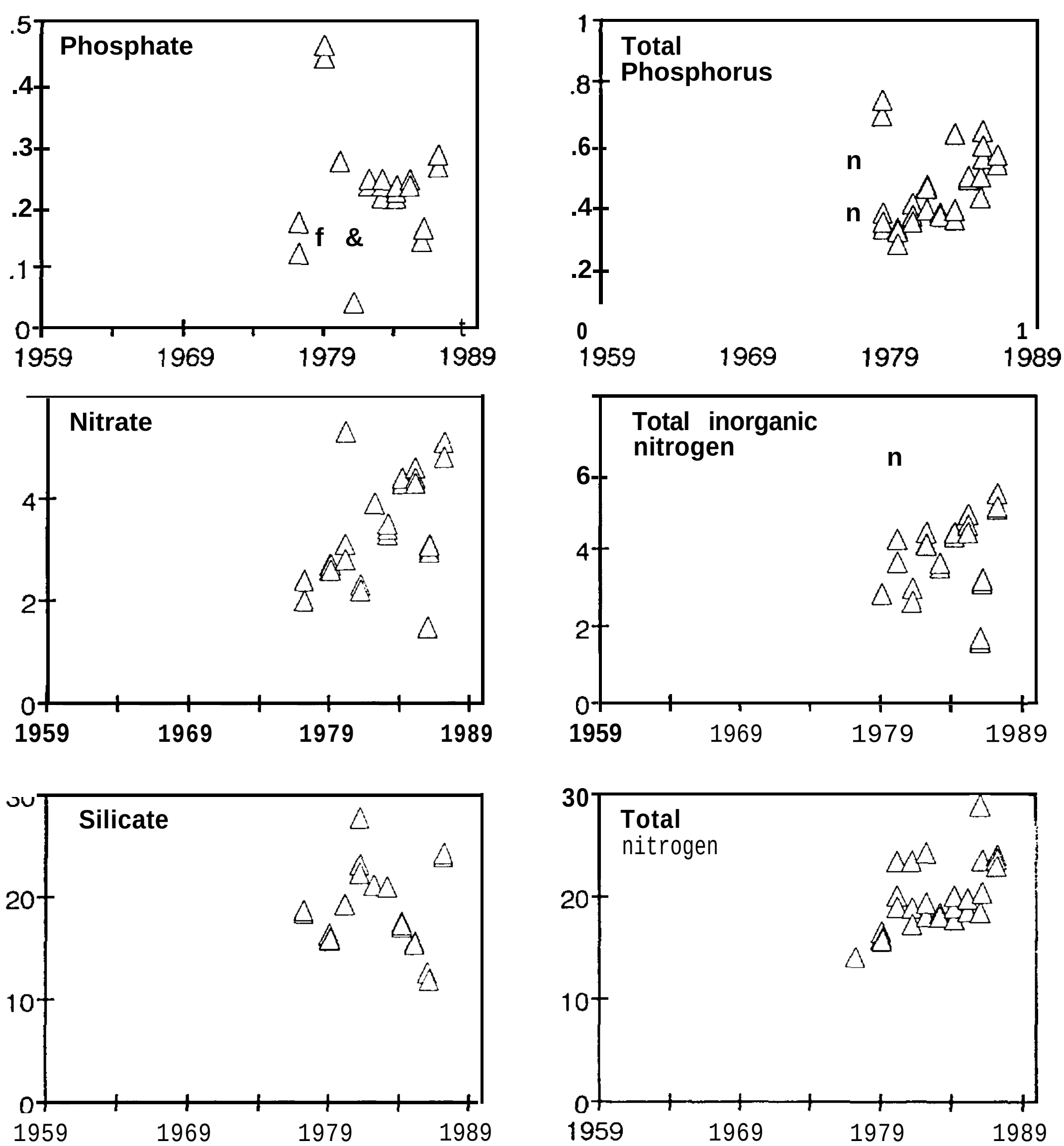

Figure 14a. Long-term variations of nutrient concentrations in the Bothnian Sea (BMP C4 = SR 5); winter concentrations (1 Jan - 31 Mar) in the surface layer (0-12 m).

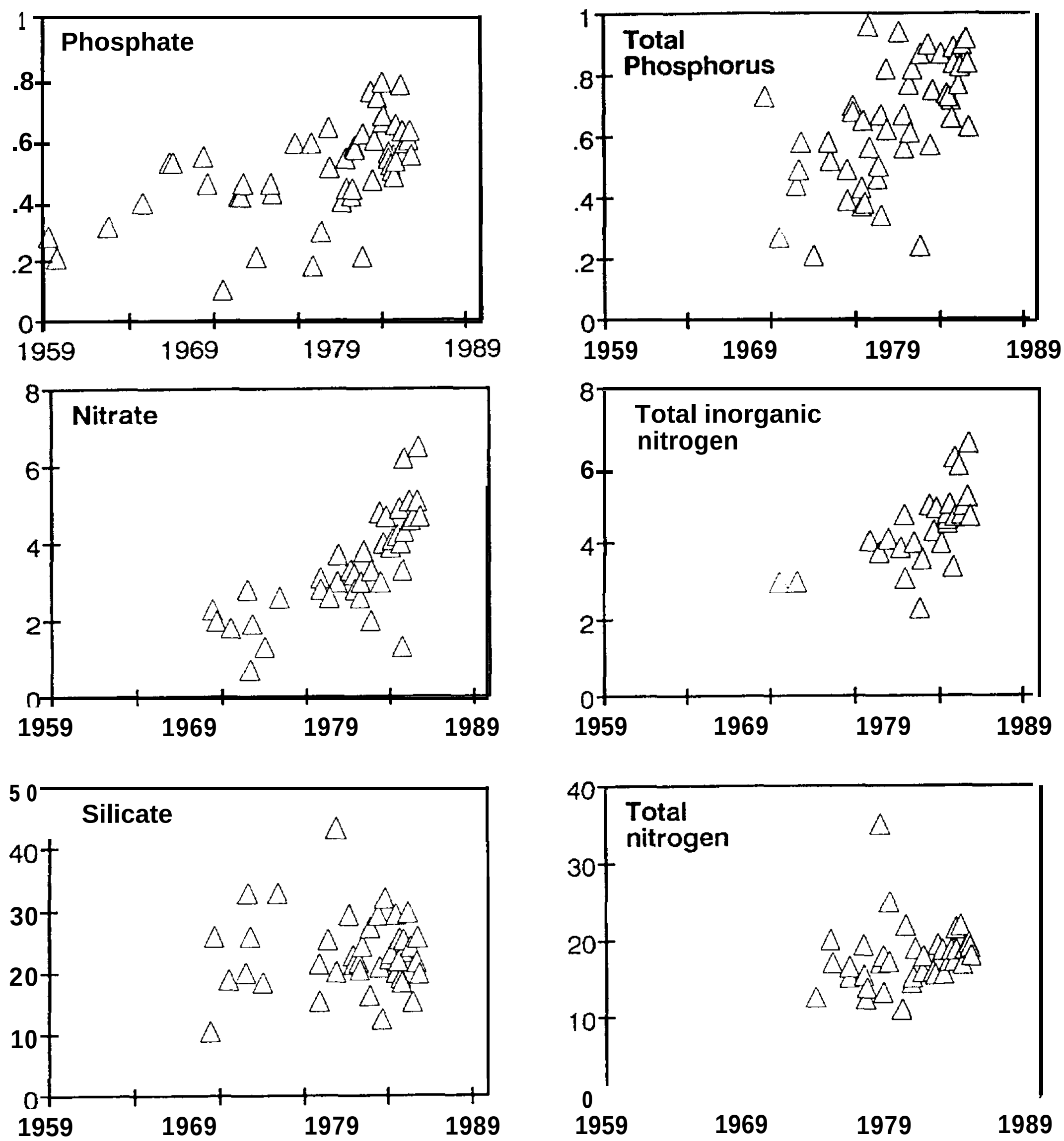

Figure 14b. Long-term variations of nutrient concentrations in the Bothnian Sea (BMP C4 = SR 5) at 100 m depth (year-round data).

Additional information on the development of the nutrient conditions in the Gulf of Bothnia is available by Perttilä (1989). The basis of his studies is the regular monitoring of hydrochemical parameters in this area carried out by the Finnish Institute of Marine Research, Helsinki, since 1966.

Perttilä could show that omitting the low summer values nitrate concentrations are increasing in the surface layer of the Bothnian Sea (Fig. 15). This trend is difficult to analyse quantitatively because of large variations owing to the irregular sampling time. However, in the deep layer of this area, the trend is evident (Fig. 16). Since the early 1970s, the deep water pool of nitrate characterised by the concentrations in 80 m depth has increased from about 2 μmol dm^{-3} to 6 μmol dm^{-3} in 1987.

The nitrate trend including data throughout the year is very clear at 30 m depth in the Bothnian Bay (Fig. 17). The mean annual accumulation rate of nitrate is 0.22 μmol dm^{-3} in this depth.

The summarised results show that except for seasonal variations phosphate concentrations have remained at the same level in the entire Gulf of Bothnia since 1978. Nitrate concentrations have increased in both the surface layer and the deep water in the Gulf of Bothnia. In the Bothnian Bay, the accumulation of this nutrient is quite steady generating a significant trend in the surface layer. The trend studies were not restricted to winter conditions but also include data of the biologically active season, because nitrate is not the limiting factor for primary productivity in this area.

3.3.7 Gulf of Finland *

No special studies on nutrient trends are available for the Gulf of Finland. Some new results covering this problem are, however, published by Kahma and Voipio (1989) and considered in the following. *)

After calculating the mean seasonal distribution using all data available for the respective nutrients in the period 1966-1988, Kahma and Voipio (1989) determined the deviations from the mean seasonal concentrations. Figure 18 shows the residuals of the mean phosphate, total phosphorus and nitrate concentrations in the layer 0-50 m at BMP Station F3 (LL7) well representing the conditions in the entire Gulf of Finland.

Whereas the increase of the phosphate concentrations was statistically insignificant, positive trends have been identified in the concentrations of total phosphorus and nitrate. The mean annual accumulation rate was 1.4 % for total phosphorus and 3 % for nitrate.

The nutrient accumulations are lower as assumed in earlier investigations in the Gulf of Finland (Baltic Marine Environment Protection Commission, 1987a). The main reason for this was identified by Voipio and Tervo (1988) in the uneven distribution of data representing the winter conditions.

*) summarized by D. Nehring

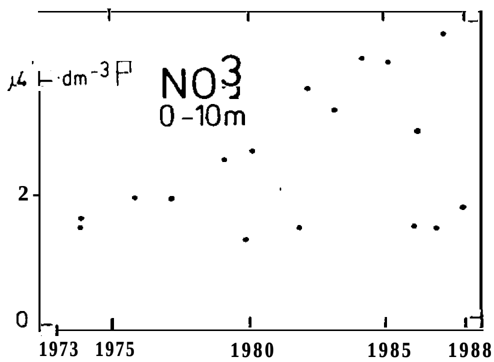

Figure 15. Winter concentrations (Dee-Mar) of nitrate in the surface layer (O-10 m) of the Bothnian Sea (BMP Cl = US 5b).

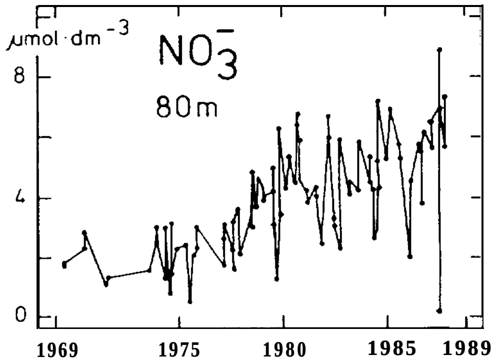

Figure 16. Nitrate development at **80 m** depth in the Bothnian Sea (BMP Cl = US 5b; year-round data).

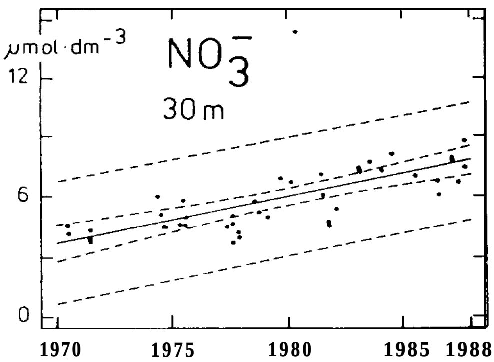

Figure 17. Nitrate trend at **30** m depth in the Bothnian Bay (BMP Al = F2; year-round data).

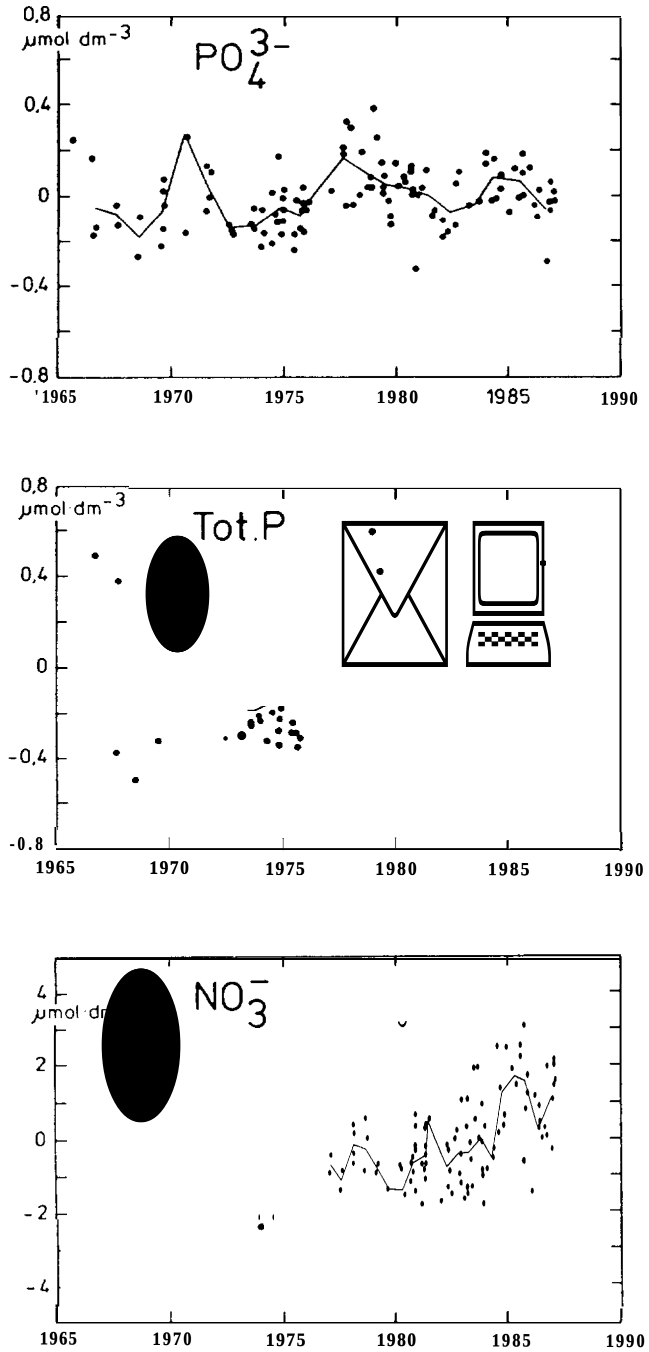

Figure 18. Trends of nutrient residuals in the Gulf of Finland (BMP F3 = LL 7; mean concentrations in the layer 0-50 m).

3.3.8 Gulf of Riga

A. Yurkovskis[5] and M. Mazmachs[5]

Studies on long-term trends have been carried out in the Gulf of Riga using the data base of the Baltic Fisheries Research Institute in Riga/USSR. Figure 19 shows the results for a central station well representing the situation in the entire Gulf.

The winter concentrations of total phosphorus and nitrate are increasing in the surface layer (Fig. 19 a, b). The nitrate accumulation is roughly 70 times higher than phosphate. This is attributed to peculiarities of the land-based nutrient discharge.

Significant positive trends in phosphorus and nitrogen concentrations were also identified in summer in both the surface layer (Fig. 19 c, d) and the deep water (Fig. 19 e, g). The long-term accumulation rate of total phosphorus in the surface layer is similar in winter and summer (Table 9).

Detailed investigations in the surface layer in summer yield reliable relationships of total phosphorus and nitrate with the river run-off and the salinity which attribute the year-to-year dynamics of nutrients to variations of the riverine water discharge. The reverse correlation between nitrate concentrations and water temperatures indicates the prevailing significance of vertical mixing through the thermocline. No long-term trend could be identified for silicate in the surface layer.

Statistic analysis shows that the river run-off during the autumn-winter-spring period and the stability of the thermocline (the temperatures in the upper layer) determine the nitrate concentrations in the deep water in summer. The ratio between the nitrate and phosphorus accumulations is higher in winter than in summer.

Recently, the silicate concentrations in summer are markedly dropping in the deep water of the Gulf (Fig. 19 f). However, the short period under investigation does not permit conclusions about long-term trends. Variations in the silicate concentrations are reversely correlated with the river run-off and directly correlated with the salinity indicating the significance of the water exchange with the Baltic Proper. Table 9 contains the mean annual accumulation rates of the nutrients studied in the Gulf of Riga.

Table 9. Mean annual accumulation rates (trend coefficients) of nutrients ($\mu mol \cdot dm^{-3}\ yr^{-1}$) in the Gulf of Riga at 57°36' N, 23°38' E.

Months	Depths/m	PO_4^{3-} 1982-1989	Tot.-P 1972-1989	NO_3^- 1975-1989	SiO_4^{4-} 1974-1989
Feb.	0-10		0.019	(0.74)*	
Aug.	0-10		0.016	0.16	
Aug.	20-50	0.024		0.49	-3.0

* 1978-1989

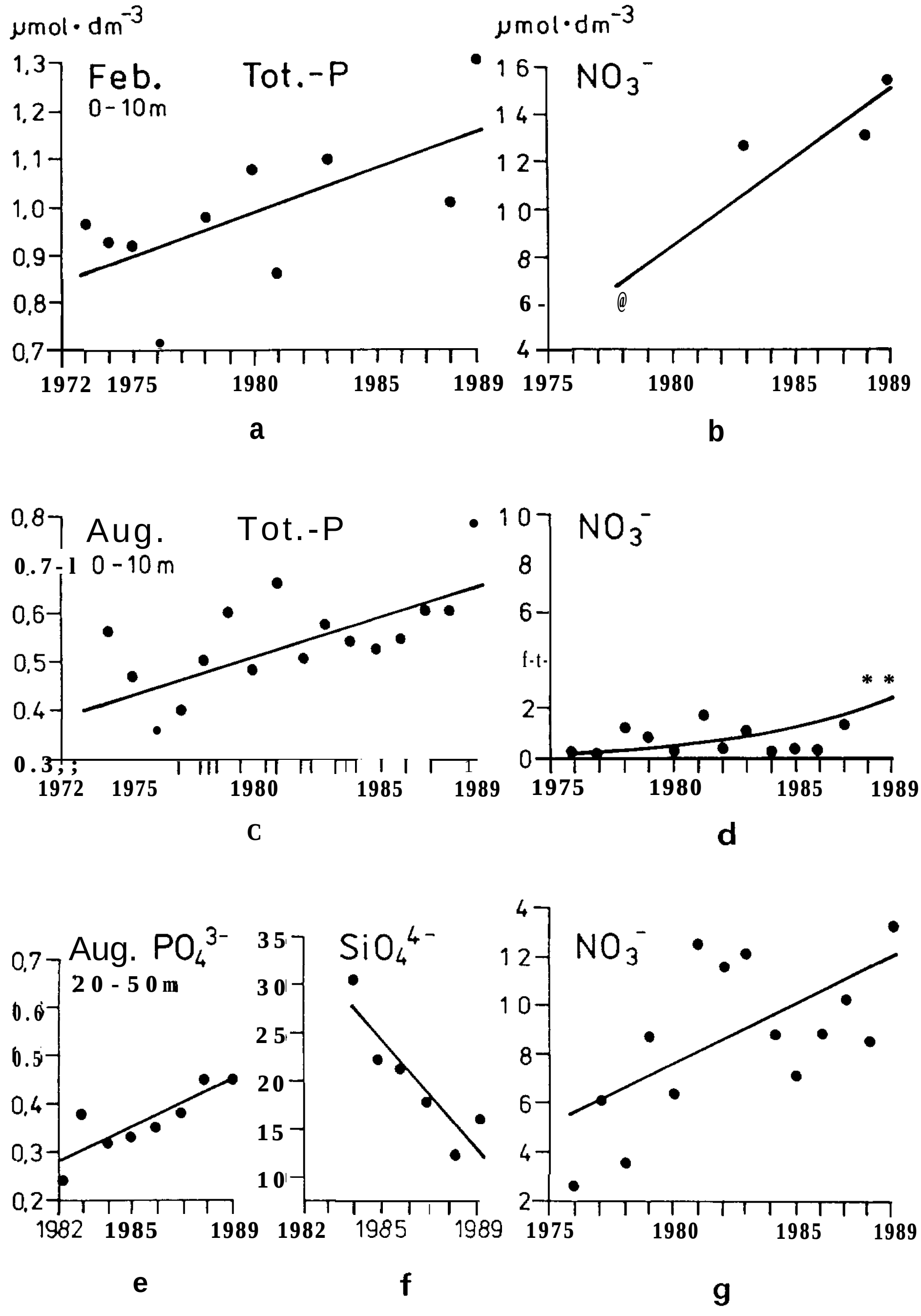

Figure 19. **Phosphate, total phosphorus, nitrate and silicate trends in the surface layer (0-10 m) and in the deep water (20-50 m) in the Gulf of Riga (Position** 57°36' N, 23°38' E).

3.4 DISCUSSION

D. Nehring[1], H.P. Hansen', A. Trzosińska[4]

Phosphorus and nitrogen compounds are the driving forces in eutrophication. Studies on long-term variations beginning in some cases in the 1950s indicated, on average, increasing winter concentrations of these nutrients in the surface layer of all sub-regions in the Baltic Sea area until 1988. These trends often result from the strong increase in the period 1969 to 1978. Thereafter the phosphate and sometimes also the nitrate concentrations remain at their relatively high level. Exceptions are the Kattegat and the Gulf of Riga characterized by a further increase of nutrient concentrations. Increasing nitrate concentrations have been observed in the Bay of Gdańsk and in the whole of the Gulf of Bothnia as well as in the Gulf of Finland.

Trend changes observed in the surface layer are also reflected to a certain extent in the oxic deep water of the Central Baltic basins.

High phosphate accumulation rates have been identified in the near-bottom water layers of the Central Baltic deeps with predominant anoxic conditions in the recent stagnation period beginning in 197611977. They mainly result from the remobilization of phosphate by reduction of the iron hydroxo complex in the sediments caused by the increasing hydrogen sulphide concentrations and consequently decreasing redox potential. Thus, the strong increase of phosphate concentrations observed under these conditions has rather natural than anthropogenic reasons, depending on missing major Baltic inflows.

Since there is no indication of considerable changes in potential land-based and airborne sources, the insignificant changes in phosphorus and nitrogen concentrations recently observed, on average, in the winter surface layer in most parts of the Baltic Sea area can be discussed as follows:

- Steady state has developed between inputs, biogeochemical sinks, and recycling of nutrients also including the nutrient exchange through the entrances of the Baltic Sea.
- Interannual 3 to 4 and 6 to 7 year cycles attributed to changes in the atmospheric circulation create variations in the river run-off and are also reflected by changes in the nutrient trends (Trzosiiiska, 1990).

The phosphorus and nitrogen concentrations included in the biogeochemical cycle are recently at such a high level in the Baltic Sea area that the sedimentation and microbial destruction of biogenic material produced in the euphotic layer cause further deterioration of the oxygen conditions and/or spreading of the anoxic zones in the deep water. Areas covered by these unfavourable developments are the Kattegat, the Belt Sea, the Arkona Sea, the Bornholm Sea and the Bothnian Sea.

Silicate concentrations in the Baltic Sea area have recently been decreasing, on average, in the surface layer. Reasons discussed in this connection are intensified diatom blooms favoured by eutrophication and subsequent sedimentation of these algae (Wulff and Rahm, 1989).

The local meteorological and hydrographic conditions, like upwelling and vertical mixing in connection with the wind field, precipitation, and river run-off strongly influence the nutrient concentrations in the winter surface layer of the more shallow western parts of the Baltic Sea area and the Gulf of Riga. They render more difficult trend studies in these regions.

SUMMARY

Winter concentrations of phosphate, total phosphorus, or inorganic nitrogen compounds are increasing in the Kattegat (1974 to 1988) and the Gulf of Riga (1971-1989). Winter concentrations of phosphate and nitrate are at a high level but do no longer increase, on average, in the surface layer of the offshore regions in the Belt Sea and the Baltic Proper in the period 1978-1989. The positive overall trends of these nutrients mainly result from the high accumulation rates in the period 1969-1977. The trends in the Arkona Sea, the Bornholm Sea and the Eastern Gotland Sea are interrupted by decreasing phosphate and nitrate concentrations from 1976 to 1980 and 1984 to 1989.

Nitrate concentrations are increasing in the winter surface layer of the Gdańsk Basin in the whole period under investigation (1970-1988). Nitrate concentrations are also increasing in the whole of the Gulf of Bothnia, whereas phosphate remained at the same level in this area since 1978.

Nutrient accumulations are lower than assumed in earlier studies in the Gulf of Finland (Baltic Marine Environment Protection Commission, 1987a). Using the deviations from the mean seasonal variations in the period 1966-1987 positive trends in the nitrate and total phosphorus residuals have been identified for the mean concentrations in the layer 0-50 m, whereas phosphate residuals increased only insignificantly, on average.

Trend studies in Central Baltic deep waters covering the period since the last major Baltic inflow in 197611977 yield lower accumulation rates for phosphate and nitrate in the oxic layer below the permanent halocline, in comparison with the previous period.

High phosphate accumulation rates have been identified in the near-bottom water layers of the Central Baltic deeps with predominant anoxic conditions in the recent stagnation period beginning in 197611977. They mainly result from the transient remobilization of phosphate by reduction of the iron hydroxo complex in the sediments caused by the increasing hydrogen sulphide concentrations and consequently the decreasing redox potential. Relations to the long-lasting recent stagnation period are obvious.

Phosphate trends are also positive in deep waters of the Bornholm Basin and Gdańsk Deep characterised by alternating oxic and anoxic conditions. The elimination of values measured in the presence of hydrogen sulphide considerably lowered the accumulation rates. The rates identified under oxic conditions reflect the phosphate accumulation caused by eutrophication.

Silicate concentrations studied in the Kattegat, the Belt Sea, the Baltic Proper, the Gulf of Bothnia, and the Gulf of Riga are decreasing, on average, in the surface layer.

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Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea, *1984-1988; Background Document*

4. PELAQIC BIOLOGY

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ABSTRACT

All data on the pelagic biology show a very high interannual variability. The present state of the Baltic Sea is determined by a continuing eutrophication process. The effect in the open sea are still moderate. The eutrophication is related to the anthropogenic nutrient input ant to the remobilized nutrient *reserves* from the deep water. During the

interannual variability which was mainly related to meteorological features. It was emphasised further that our understanding on the consequences of the stagnation period which begun in 1976 is much too small for a comprehensive interpretation of observed phenomena.

4.2 MATERIAL AND METHODS

This is the first attempt to use the plankton data in the HELCOM data base to assess the state of the Baltic Sea. In addition to the HELCOM data, some countries have appended national data in order to expand the temporal range of the data set and to increase the frequency of the observations.

The samples for phyto- and zooplankton species composition and biomass, chlorophyll-a, as well as primary production capacity (P_{max}, incubator) and daily production determinations (in situ-incubation) were collected in 1979-1988. The sampling and analysis were carried out according to the Guidelines for the BMP (Baltic Marine Environment Protection Commission, 1984). Means of O-10 m are used for phytoplankton variables.

The data is processed divided into geographical subareas and into growth seasons. In general, data from all sampling stations inside a geographical area is pooled.

In the Danish data (Kattegat, Sound, and Great Belt) values within July-September/early October have been averaged to give a mean annual summer value, which have then been used for the analysis. A few very outstanding results have been excluded in the analysis. In the area between Kattegat and the Gotland Sea the period May-June is selected to represent the post bloom season, and July-September to represent the summer season for chlorophyll a and primary productivity data. In the Northern Baltic Proper, in the Gulf of Finland as well as in the Gulf of Bothnia, the period June-September represents summer for chlorophyll a and primary productivity data. The data for the spring period appeared to be too scarce for further analysis. In the Bothnian Bay the annual phytoplankton maxima coincide with the period June-September.

Data for phytoplankton species composition and biomass was pooled in the following way: spring comprises March-April in Kattegat, Belt Sea and The Sound and April-May in all other areas; for Bothnian Sea and Bothnian Bay also June has been added. Summer is July, August, and September and autumn the rest of the year. The phytoplankton data represent samples from the depths of 0 m or O-10 m (more than 25 000 observations). The total data set received from HELCOM contained some obvious errors in the calculations of total volumes, etc., mainly caused by mistakes in the data reporting. These wrong values have been omitted or corrected when it was possible. After these corrections 2 962 quantitative observations had no given mean volumes of species. In most cases, it was possible, however, to use a median volume figure for the species in question and in this way nearly the whole data set could be completed.

For zooplankton January-March represents winter stage, April-June spring, July-September summer, and October-December autumn. In the Gulf of Finland and in the Gulf of Bothnia data from the period July-September, representing the summer, was used.

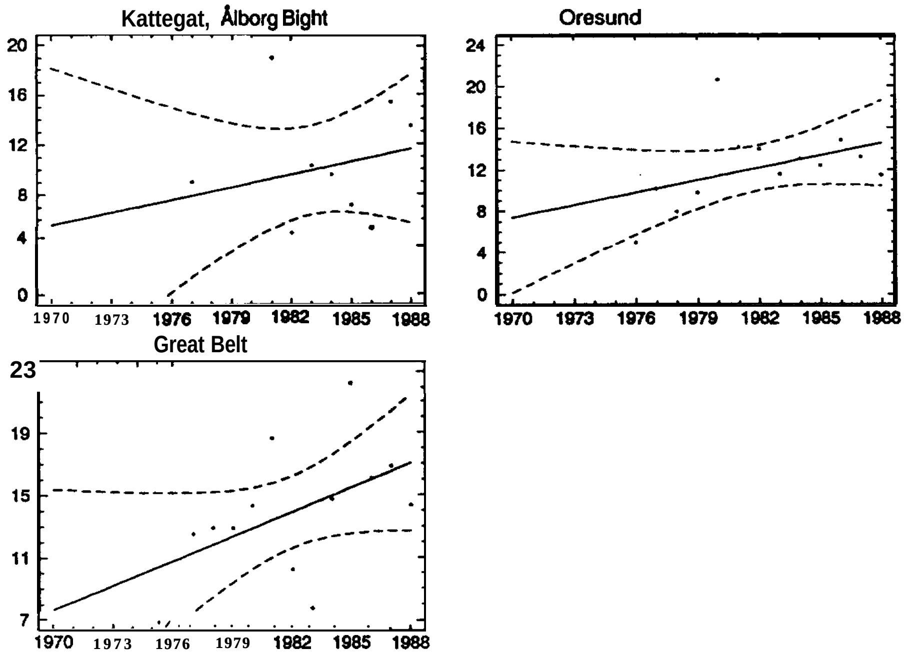

Figure 2. The annual means of primary production capacity ($mgCm^{-3}h^{-1}$) at sampling stations representing Kattegat, Sound (Öresund) and Great Belt. The solid line represents linear regression between the annual mean values and year, the broken ones represent 95% confidence limits for the regression line.

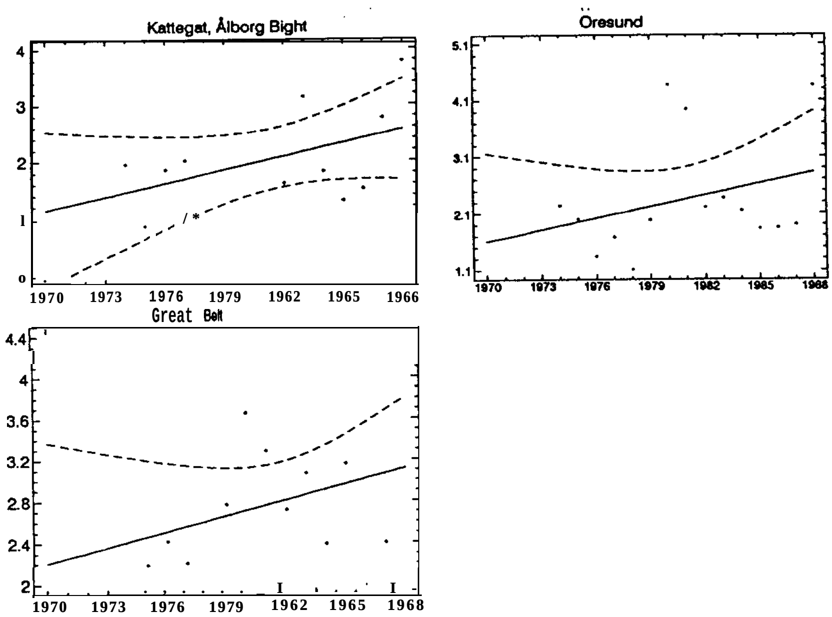

Figure 3. The annual means of chlorophyll a ($mg\ m^{-3}$) at sampling stations representing Kattegat, Sound (Öresund) and Great Belt. The solid line represents linear regression between the annual mean values and year, the broken ones represent 95% confidence limits for the regression line.

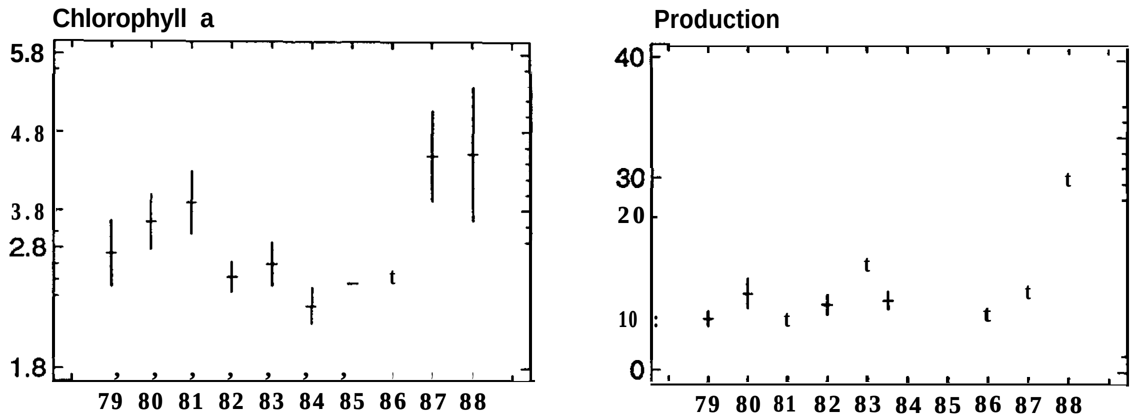

Figure 5. *The* annual means of chlorophyll a *(mg* m^{-3}) and primary production capacity (mgC m^{-3} h^{-1}) at the sampling stations representing Kiel Bay during the period July-September. The vertical bars represent standard error.

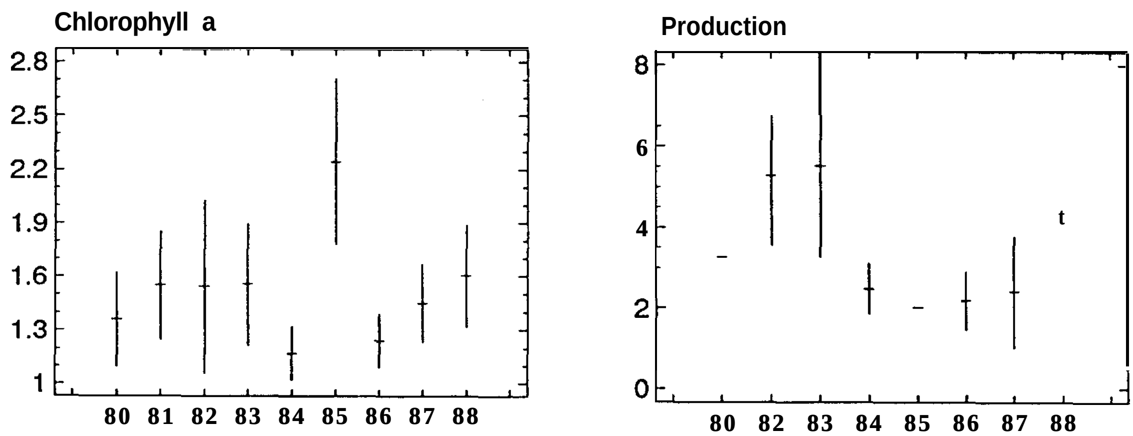

Figure 6. The annual means of chlorophyll a (mg m^{-3}) and primary production capacity (mgC m^{-3} h^{-1}) at the sampling stations representing Bay of Mecklenburg during the period May-June. The vertical bars represent standard error.

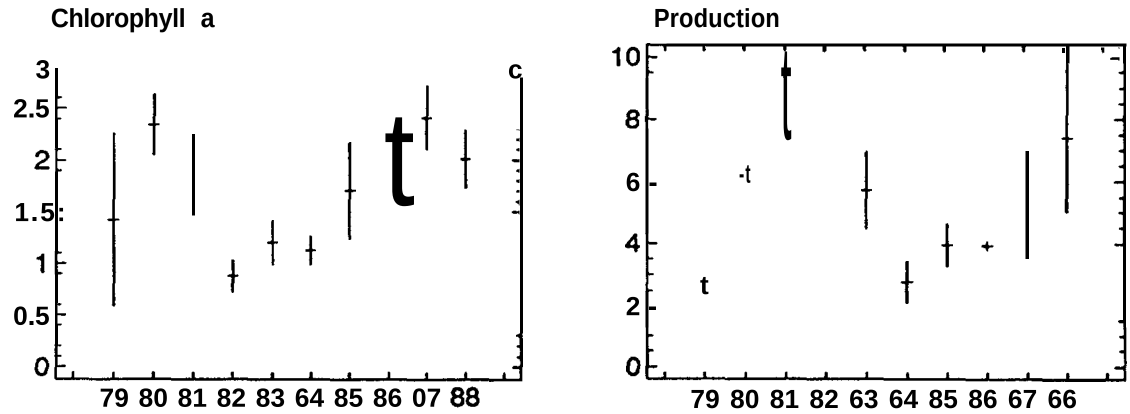

Figure 8. The annual means of chlorophyll-a (mg m^{-3}) and primary production capacity (mgC $m^{-3}h^{-1}$) at the sampling stations representing Arkona Sea during the period May-June. The vertical bars represent standard error.

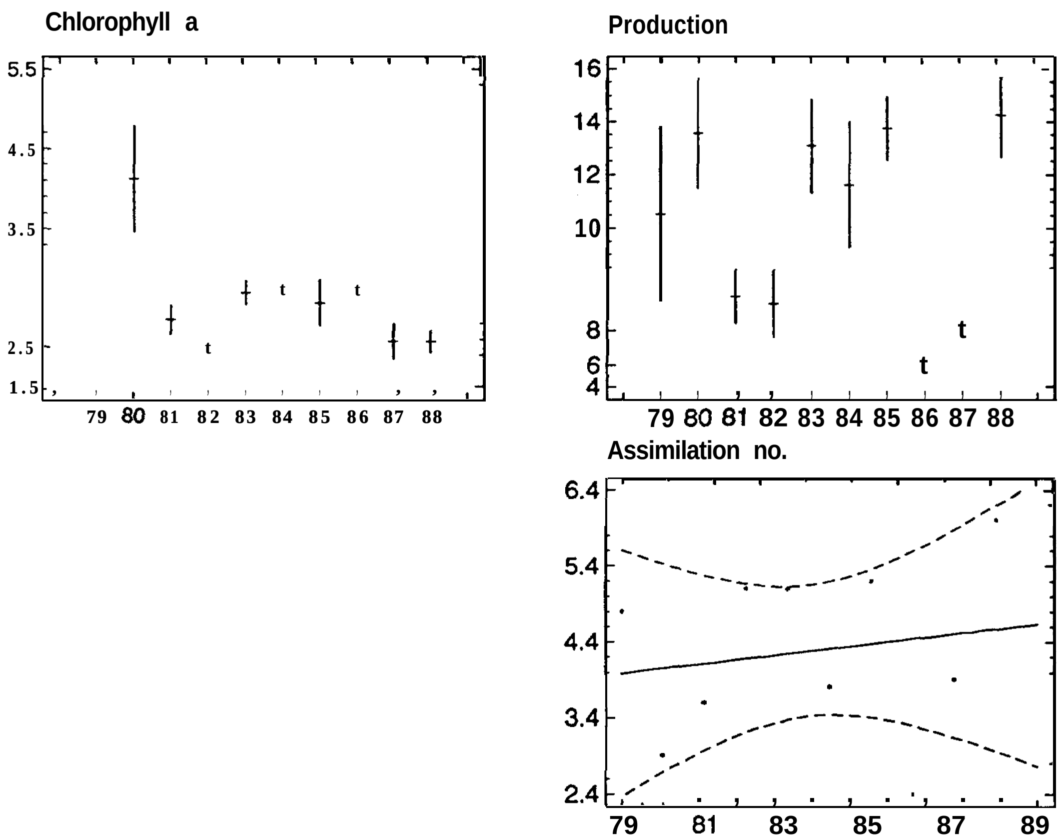

Figure 9. The annual means of chlorophyll-a (mg m^{-3}), primary production capacity (mgC $m^{-3}h^{-1}$) and assimilation number (mgC $m^{-3}h^{-1}$/ mg Chl m^{-3}) at the sampling stations representing Arkona Sea during the period July-September. The vertical bars represent standard error. The solid line is the regression between assimilation number and year and the broken lines show the 95% confidence limits.

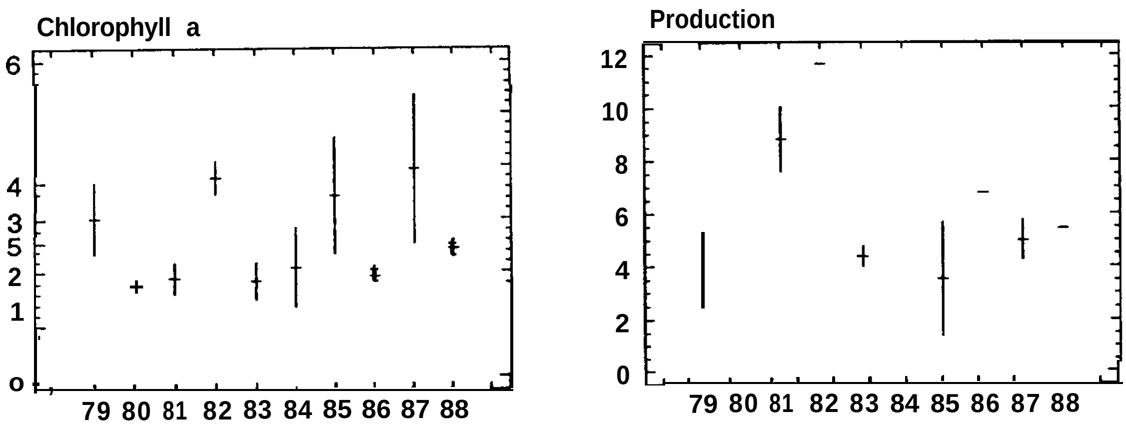

Figure 10. The annual means of chlorophyll a (mg m^{-3}) and primary production capacity (mgC m^{-3} h^{-1}) at the sampling stations representing Bornholm Sea during the period May-June. The vertical bars represent standard error. The solid line is the regression between assimilation number and the year and the broken lines show the 95% confidence limits.

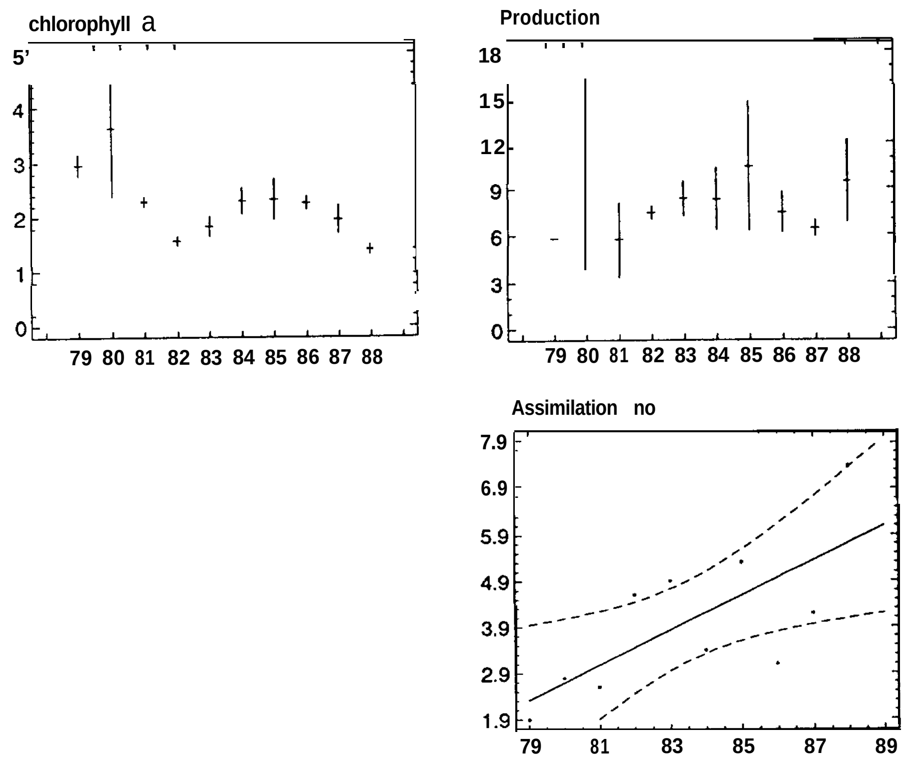

Figure 11. The annual means of chlorophyll a (mg m^{-3}), primary production capacity (mgC m^{-3} h^{-1}) and assimilation number (mgC m^{-3} h^{-1}/ mg Chl m^{-3}) at the sampling stations representing Bornholm Sea during the period July-September. The vertical bars represent standard error. The solid line is the regression between assimilation number and the year and the broken lines show the 95% confidence limits.

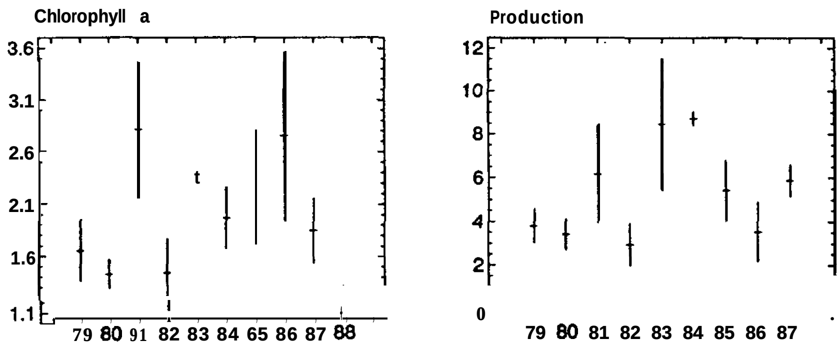

Figure 14. The annual means of chlorophyll a (mg m^{-3}) and primary production capacity (mgC $m^{-3}h^{-1}$) at the sampling stations representing Northern Baltic Proper during the period June-September The vertical bars represent standard error.

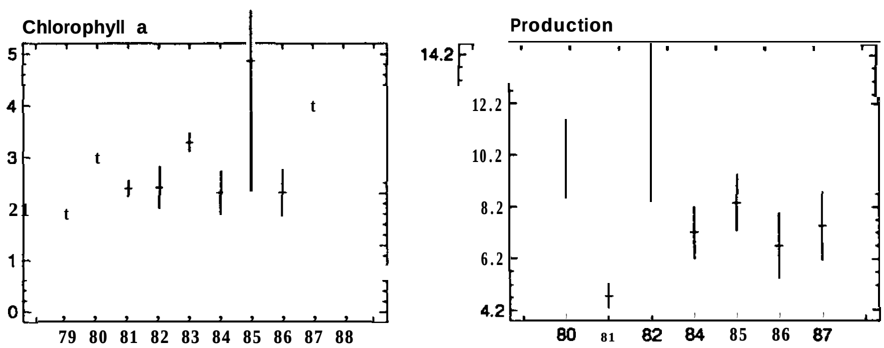

Figure 15. The annual means of chlorophyll a (mg m^{-3}) and primary production capacity (mgC $m^{-3}h^{-1}$) at the sampling stations representing the Gulf of Finland during the period June-September The vertical bars represent standard error.

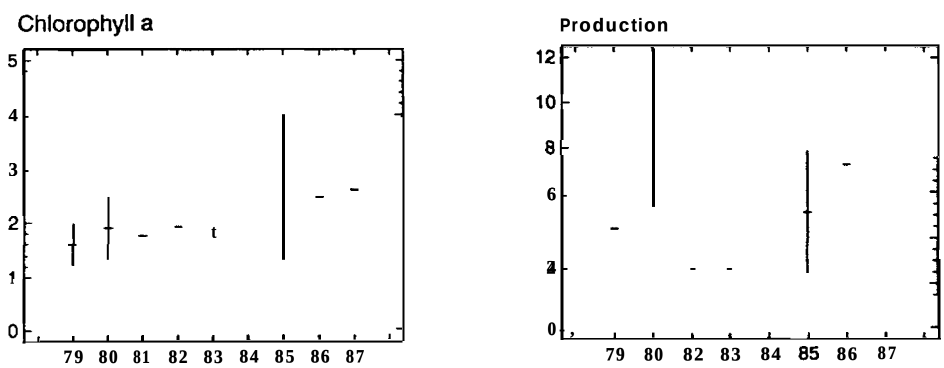

Figure 16. The annual means of chlorophyll a (mg m^{-3}) and primary production capacity (mgC $m^{-3}h^{-1}$) at the sampling stations representing the Åland Sea during the period June-September The vertical bars represent standard error.

4.4 PHYTOPLANKTON BIOMASS AND SPECIES COMPOSITION

T. Willén[4], K. Kononen[2] and U. Horstmann[3]

4.4.1 The Kattegat

Reference station: 413 Anholt (BMP R3)

The spring period (Fig. 19) is quite dominated by diatoms reaching peak values in 1983 and 1984. The predominating taxa - in this case ***Detonula confervacea*** and ***Porosira glacialis*** - are given in Table 1. In summer, several algal groups contribute to a total biomass only exceeding 4 mg/l in 1980 when cyanobacteria, mainly ***Gomphosphaeria*** sp. with 3.7 mg/l, together with other groups made up a total of 7.5 mg/l. Dinoflagellates, above all ***Ceratium tripos, C. furca, C. lineatum*** and ***Prorocentrum micans,*** played a prominent role both in summer and autumn; no real peak values, however, are observed in the material.

At Anholt extremely high total biomass values were received in October 1981 (mainly diatoms), in March 1983 (diatoms) and in March 1986 (diatoms). Cyanobacteria were found in quantities worth mentioning only in summer 1982 while Dinophyceae has shown an increasing trend from 1986. Comparing total biomass values from the two assessment periods no significant change was observed (Mann-Whitney test). Spring values from the whole period show a slight (not significant) increase; the autumn values, however, show a significant increase during the period 1979-1988 (one maximum value deleted).

From the Kattegat-Skagerrak area several occasions of mass development of various species have been reported, e.g., toxin producing dinoflagellates, ***Chrysochromulina polylepis, Gyrodinium aureolum,*** etc.

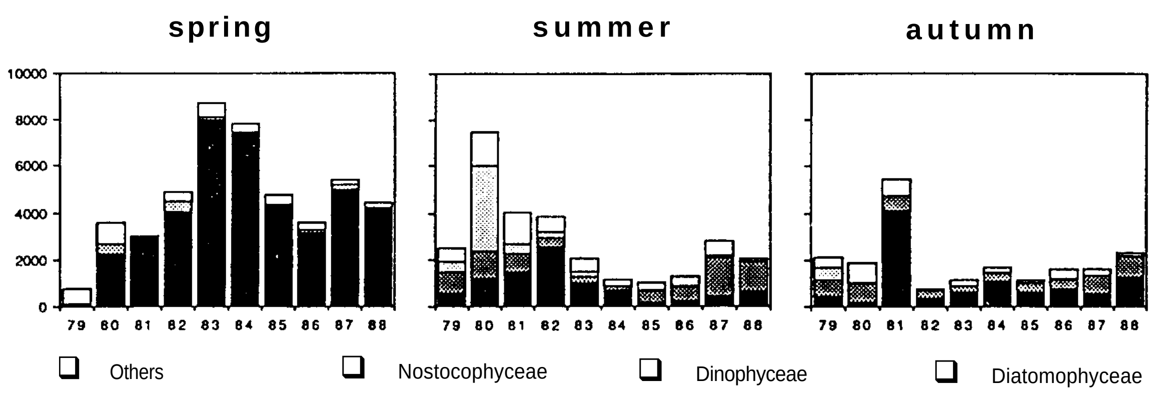

Figure 19. Distribution of the different algal classes in spring, summer and autumn 1979-1988 in Kattegat (413 Anholt = BMP R3). Mean values in µg/l.

'5.5.2 The Belt Sea

<u>Reference stations</u>: Halsskov (BMP Pl), Fehmarnbelt (BMP N1)

In spring (Fig. 20) there is a total dominance of diatoms *(Detonula, Thalassiosira, Skeletonema* and *Achnanthes;* Table 1) with the highest total volume value in 1979. Two very low diatom values were observed in 1987 and 1988. In both cases the bloom was missed in the observation. In 1987 it was over and in 1988 it has not yet started. Highly fluctuating values were observed in summer: the highest in 1983, the lowest *in 1986.* In some years a considerable development of cyanobacteria *(Aphanothece, Gomphosphaeria)* was observed; in 1988 dinoflagellates *(Ceratium tripos)* showed a maximum value. The autumn development do not show any marked differences during the whole period.

At Halsskov sometimes very high single total volume values were obtained during the period 1979-1985. After 1985 no peak records were received and there is a tendency to decreasing biomass. The highest values of cyanobacteria were observed in August 1979 and 1980; the dinoflagellates reached a total biomass more than 1.5 mg/l only in 1986 and the diatoms showed two maximum values: in October 1981 and in March 1985. Comparing the two assessment periods, a significant decrease of the summer values at Halsskov has been observed.

In general, the Fehmarnbelt values were smaller than corresponding values from Halsskov. No significant difference was obtained when comparing the total biomass values from the two assessment periods. In July 1984 the blue-green algae reached their maximum; in most summers values of 0.5 mg/l and less were observed. The dinoflagellates showed a very limited development in 1981-1985 but a slight increase from 1986 with two peak values: one in December 1986 and another in July 1988. The diatom development was rather slight during the whole period but showed a few peak values: in July 1981 and in March 1985 and 1986. Other groups, mainly small flagellates, increased slowly in total volume from 1979 to 1987. No significant changes in the seasonal development of any group could be verified.

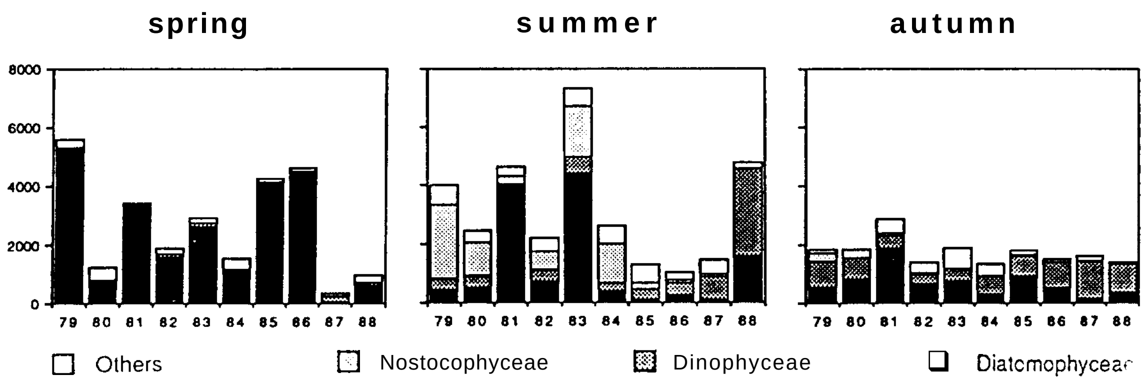

Figure 20. Distribution of the different algal classes in spring, summer and autumn 1979-1988 in the Belt Sea. Mean values in $\mu g/l$.

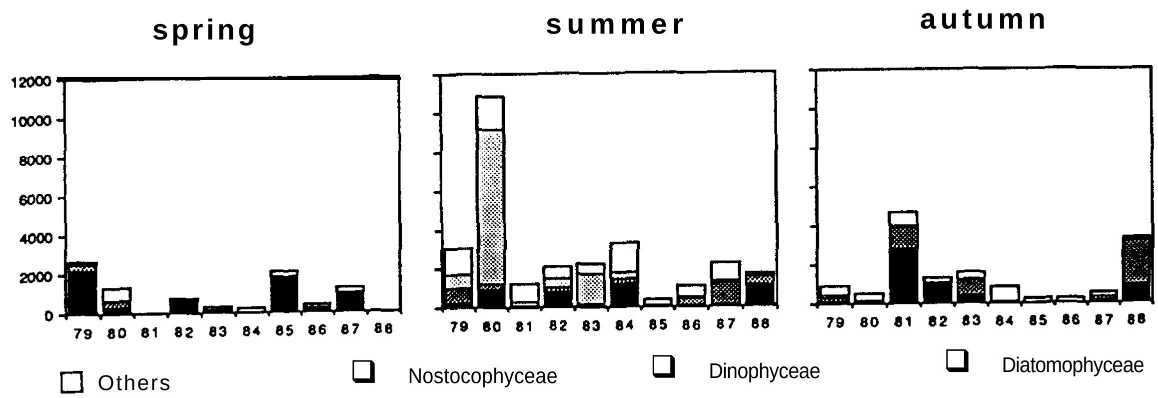

Figure 21. Distribution of the different algal classes in spring, summer and autumn 1979-1988 in the Sound. Mean values in $\mu g/l$.

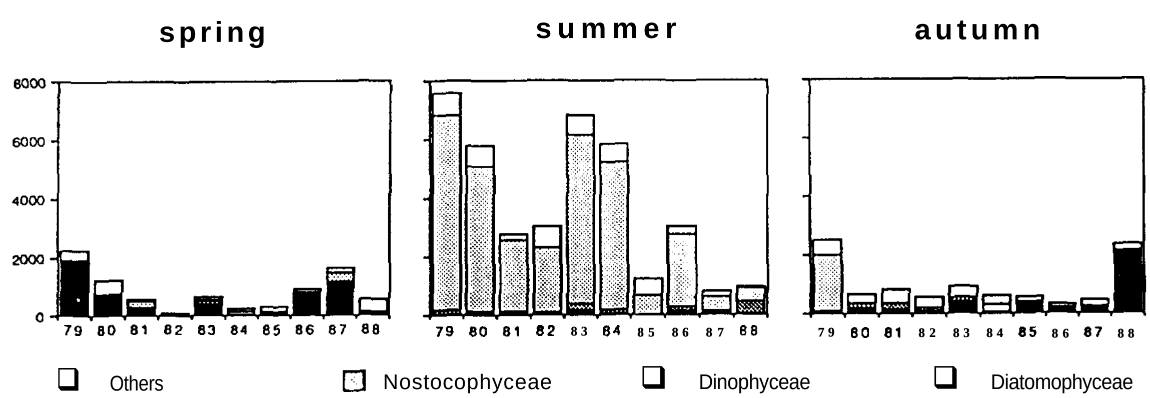

Figure 22. Distribution of the different algal classes in spring, summer and autumn 1979-1988 in the Arkona Sea (BY 2 = BMP K4). Mean values in $\mu g/l$.

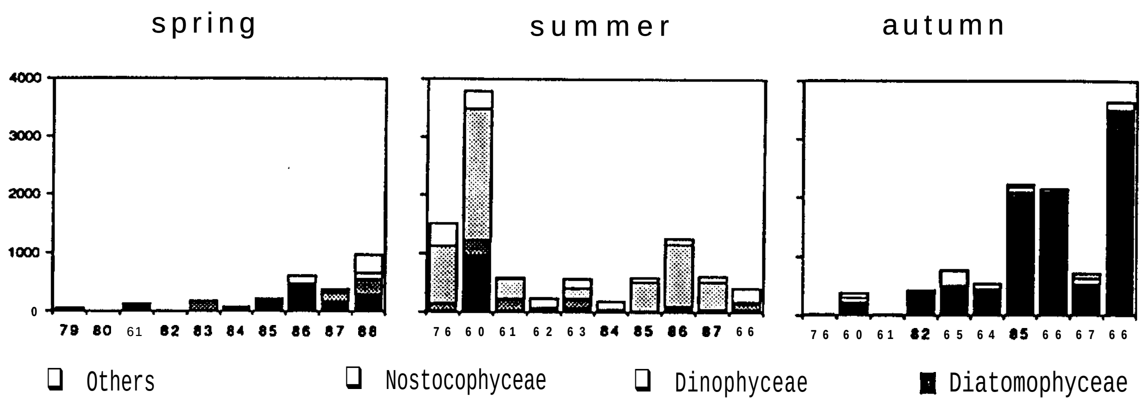

Figure 23. Distribution of the different **algal classes** in spring, summer and autumn 1979-1988 in the Bornholm **Sea** BY 5 = BMP K2). Mean values in $\mu g/l$.

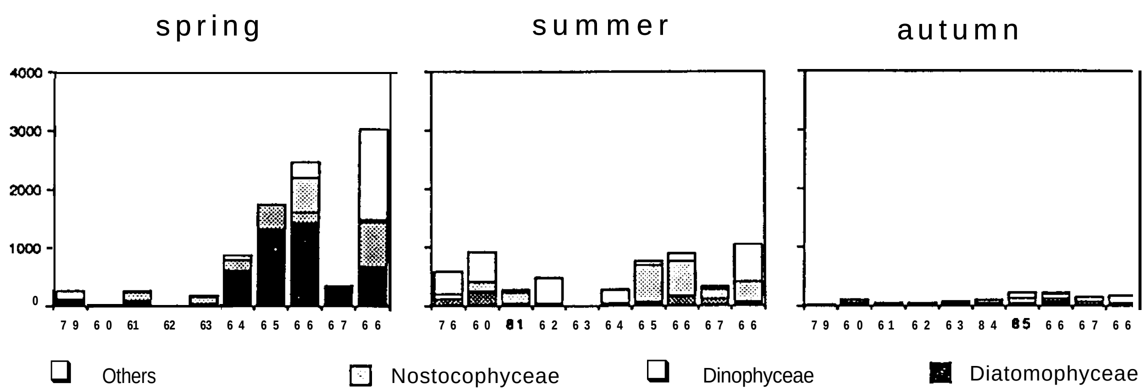

Figure 24. Distribution of the different algal classes in spring, summer and autumn 1979-1988 in the Western **Gotland** Sea (BY 31 = BMP H3). Mean values in $\mu g/l$.

4.4.8 Gulf of Finland

Reference station: LL 7 (BMP F3)

The dominating phytoplankton species during the spring are presented in Table 1. The total biomass as well as the proportions of the dominating algal groups - Dinophyceae and Bacillariophyceae - varied between years and areas. For example in 1985 the spring bloom in the middle of the Gulf of Finland was dominated with almost equal proportions by dinoflagellates and diatoms whereas in the western part the blooms was entirely dominated by diatoms.

The dominating phytoplankton groups during summer sampling were blue-green algae, dinoflagellates and flagellates (Table 1, Fig. 26). ***Aphanizomenon flos-aquae, Cryptomonas spp.*** and unidentified flagellates dominated during every summer in all parts of the area. Other species mentioned in Table 1 had more occasional dominance. The species composition is in accordance to earlier studies of phytoplankton succession in the area (e.g. Niemi 1973, 1975, Kononen and Niemi 1984). Only one species, ***Dinophysis norvegica,*** had a clear increase from the period 1979-1984 to the period 1985-1988. The total biomass values varied between years remarkably. No clear trends of any kind could be demonstrated. In 1985 the amount of blue-green algae, mostly ***Aphanizomenon flos-aquae*** and the toxic ***Nodularia spumigena,*** was especially high. Only slight, statistically non significant, differences, between different parts of the area were found. The lowest total biomass values as well lowest biomass of Dinophyceae were generally found in the west.

According to long-term phytoplankton studies of Kononen and Niemi (1984) some marine diatoms increased in abundance during the 1970s at the entrance to the Gulf of Finland. Kononen (1988) found that some nanoplanktic species as well as a heterotrophic species increased in abundance during summer months in 1972-1985 following trends in total phosphorus level. These trends could not be demonstrated in the BMP-data probably owing to differing sampling times. The trends and changes observed by Ilus and Keskitalo (1987) and Viljamaa (1988) in phytoplankton long-term studies carried out in coastal areas exposed to thermal or municipal load were not observed in the open sea areas.

Toxic blooms of blue-green algae, especially ***Nodularia spumigena*** have frequently been observed in the Gulf of Finland (Sivonen et al., 1989a, 1989b). In October 1987 an exceptionally strong bloom of ***Microcystis aeruginosa,*** then non toxic, was built up in the eastern part of the sea area (Niemi, 1988).

4.4.9 Archipelago Sea and Gulf of Bothnia

Reference station: F 64 (BMP Dl) in the Archipelago Sea

The phytoplankton biomass during spring was dominated by diatoms and dinoflagellates (Table 1, Fig. 27). The total biomass showed a clear rising trend during the study period. The increase from the period 1979-1984 to 1985-1988 was highest and also statistically significant (+-test) in the group consisting mainly of flagellates. Also the biomass of several spring bloom diatoms like *Achnanthes taeniata, Skeletonema costatum* and *Thalassiosira baltica* increased, though not as clearly.

The summer biomass were dominated by the blue-green algae, dino-flagellates and flagellates. Slight increase was observed in the total phytoplankton biomass during the study period. This was mainly due to the increase of dinoflagellates and other flagellates. Exceptionally high amounts of the toxic *Nodularia spumigena were* observed in 1986.

Kippo-Edlund and Niemi (1986) found that the phytoplankton total biomass had increased and the species composition had become more marine from the period 1966-1970 towards 1979-1982. The results obtained from the BMP-data suggest that the increasing trend has continued during the 1980s.

Reference station: SR 5 (BMP *C4)* in the Bothnian Sea

The spring bloom in the area was dominated by dinoflagellates and diatoms (Table 1, Fig. 28). There was a clear increase in total biomass from 1979 to 1988. This trend was primarily due to intensified blooming of diatoms and flagellates. However, no rising tendencies were found for any single diatom species. *Skeletonema costatum, Chaetoceros wighamii* and *Achnanthes taeniata* had their maximum occurrences during different years.

The summer phytoplankton was dominated by flagellates and blue-green algae (Table 1) in the southern part of the area and dinoflagellates in the northern part. No tendencies of any kind could be demonstrated.

Reference station: BO 3 (BMP A3) in the Bothnian Bay

The spring bloom was dominated by diatoms and dinoflagellates (Table 1, Fig. 29). There was an increasing tendency in the amount of flagellates in the northern part of the area. The total biomass was varying.

Blue-green algae together with flagellates and dinoflagellates dominated during summer. The total biomass, proportions of algal groups or single species did not show any trends.

4.5 NUISANCE PLANKTON ALGAE

Increasing attention has been drawn to exceptional phytoplankton blooms during the last decade. Several dramatic blooms of toxic marine and brackish water species have occurred killing pelagic organisms as well as benthic fauna and flora and disturbing the marine ecosystem. The toxin producing organisms belong to various algal classes, mainly Dinophyceae and Nostocophyceae but also to Chrysophyceae, Prymnesiophyceae and Diatomophyceae. In the last mentioned class only one species, ***Nitzschia pungens,*** is known as toxic from Canada but it also occurs in the Kattegat-Arkona area.

A review of water blooming organisms is already given in the First Periodic Assessment.

The present material, as a rule sparsely collected, can not give an adequate picture of phytoplankton bloom situations: single high volume values do not give information about the length of the blooming period. Only from stations with frequent sampling and by special local investigations reliable results may be obtained.

In a short survey the following nuisance plankton algae may be mentioned.

4.5.1 Nostocophyceae

Anabaena flos-aquae is distributed from Arkona Sea to the Bothnian Bay. It is not reported to produce neurotoxins except in freshwater lakes, where also other ***Anabaena spp.*** are known as toxin producing. The same applies to ***Microcystis aeruginosa*** (Niemi 1988) and ***Oscillatoria agardhii*** which mainly develop in eutrophic coastal waters.

Aphanizomenon flos-aquae occurs in the whole area; it is not known as toxin producing except in fresh waters.

Nodularia spumigena, however, generates high blooms some years. Its hepatotoxin, nodularin, is recently characterized by Eriksson et al. (1988) and by Sivonen et al. (1989a and 1989b). Several recent cases of death of dogs are reported from Denmark (Lindström 1976), the southern coast of Sweden and Gotland (Lundberg et al. 1983).

4.5.2 Dinophyceae

Some of the following species produce toxins which can be concentrated by shellfish and then eaten by people, resulting in diseases termed DSP and PSP. The problems are mainly concentrated to the Danish, Norwegian and Swedish western coasts and not to the Baltic Sea. Some species also cause killing of fish and benthic organisms. A survey of dinoflagellate blooms is given by Graneli (1986). The volume Toxic Marine Phytoplankton (Graneli et al., 1990) presents summaries and new aspects on the problems. Nordberg (1989) argues that toxin producing algae ***(Gymnodinium catenatum)*** earlier have formed immense water blooms in Kattegat: resting spores of G. ***catenatum*** are found in sediments about 2 000 years old. Maximum values were reached 300-400 years ago and shortly after that the species disappeared.

Chrysochromulina polylepis and *Chrysochromulina spp.* This *genus* is closely related to *Prymnesium. The* toxic principles of *Prymnesium parvum* has been thoroughly studied. The toxin, hemolysin, has a membrane-disrupting effect and the same is found for the toxin or toxin complex of *Chrysochromulina polylepis.* The harmful bloom mainly in the Skagerrak area during May-June 1988 initiated numerous special studies and the results are given in a number of publications, e.g., Rosenberg et al. 1988, Estep and MacIntyre 1989, Aksnes et al. 1989, Lindahl and Rosenberg 1989, Lindahl and Dahl 1990.

From the present investigation it is evident that *Chrysochromulina spp.* have occurred during the whole investigation period with maximum values in Kattegat. Single high values were observed 1984 and 1985 at Vinga with ca 4 million cells/l, ca 5 million cells/l at Fladen (2 m level), 8 million cells at Aalborg (5 m level) and ca 11 million cells/l at Anholt (5 m level). At the routine monitoring sampling on May 24-27, 1988, rather low cell numbers were recorded compared to results of sometimes 80-100 million cells/l received by the numerous special investigations performed during May and June, when also the *Chrysochromulina* distribution at different levels was recorded.

4.6 ZOOPLANKTON

G. Behrends[3], M. Viitasalo[2], G. Breuel[1], E. Kostrichkina[6], O. Sandström[4], F. Møhlenberg[5] and P. Ciszewski[7]

4.6.1 The Kattegat

In the shallow western Kattegat the copepod species *Paracalanus parvus* dominates followed by *Pseudocalanus minutus elongatus, Oithona similis, Centropages hamatus* and carnivorous cladocerans and meroplankton. In the deeper eastern Kattegat species diversity is lower with dominance of *Pseudocalanus minutus elongatus* followed by *Paracalanus parvus, Centropages hamatus, Oithona similis,* cladocerans and *Acartia spp.* In autumn, the cladocerans disappear, but the carnivorous element remains due to large increases in the biomass of *Oithona similis.* In certain years species typical for the North Sea/Skagerrak, as *Calanus finmarchicus* and *Centropages typicus*, appear in the Kattegat, especially in the western part during spring/early summer.

A decrease in the biomass of *Acartia spp.* was detected in the period 1979 - 1988. Trends in other species, total biomass or trophic structure could not be observed (Figs. 30a-b and 31a-b).

4.6.2 The Sound, the Great Belt and the Southern Belt Sea

Species composition in the surface waters of the Sound closely reflects the composition in the Arkona Sea with dominance of *Acartia spp.* followed by *Pseudocalanus minutus elongatus, Centropages hamatus* and *Oithona similis,* but the cladocerans are not as numerous.

A highly significant decrease of *Acartia* spp. could be observed. Other trends were not detectable in the time series from 1979 - 1988 (Figs. 30c and 31c).

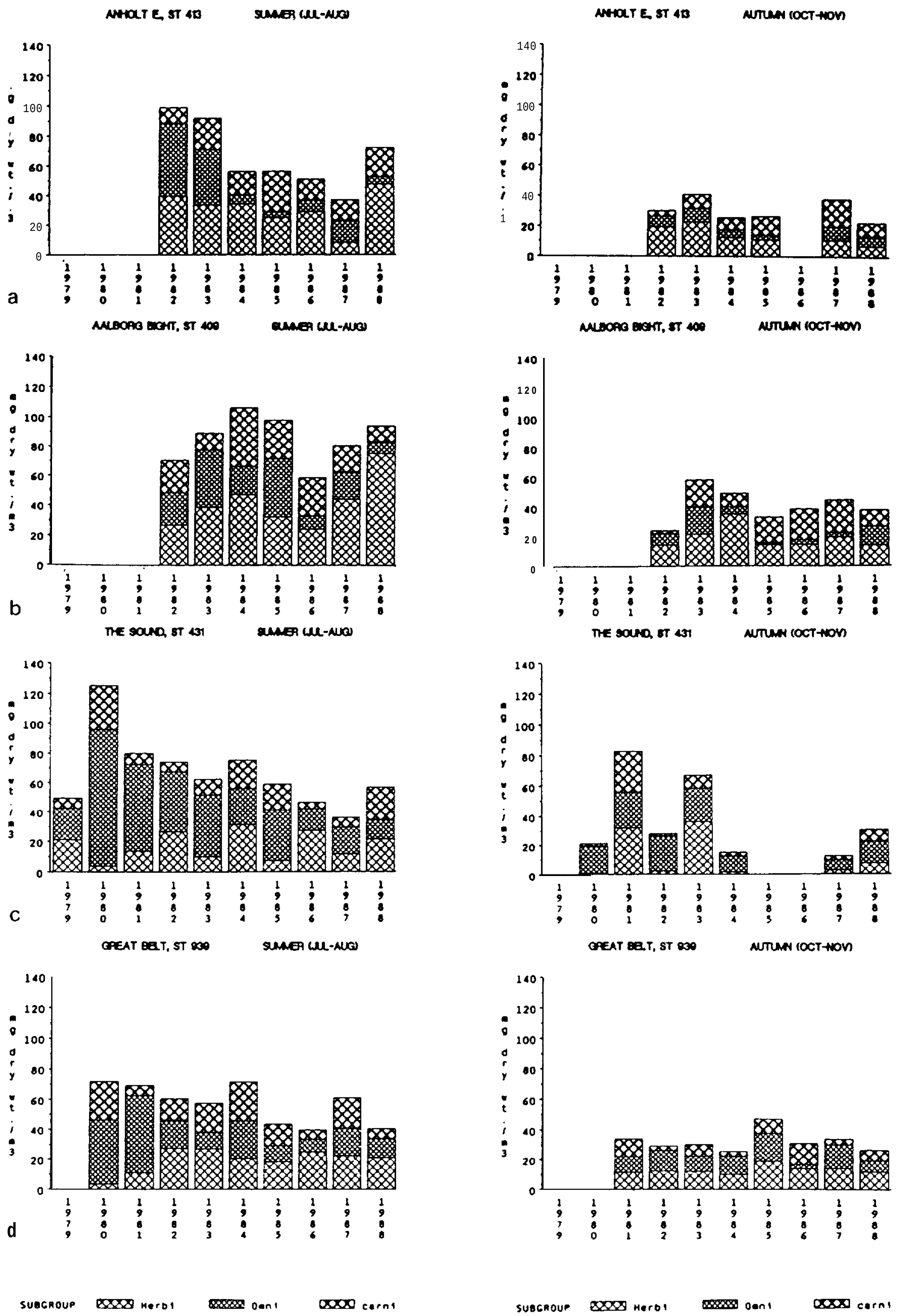

Figure 31. The abundances of zooplankton species belonging to different trophic levels in the Kattegat, the Sound and the Great Belt in different seasons.

a. station 413 = BMP R3 b. station 409 = BMP R4

c. station 431 = BMP Q2 d. station 939 = BMP P1

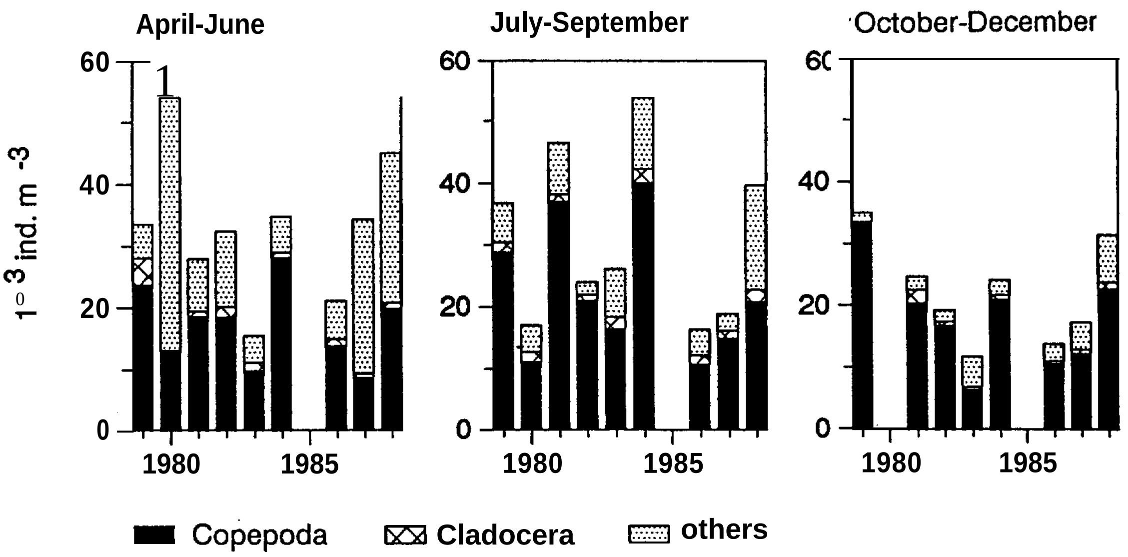

Figure 32. The abundances of the main taxonomic groups in the Belt Sea zooplankton during different seasons.

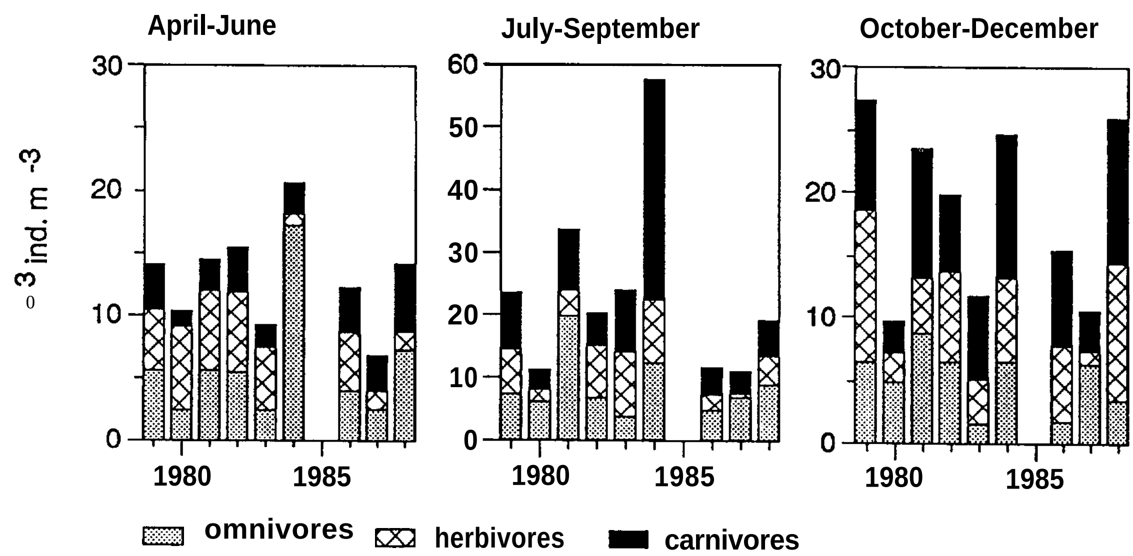

Figure 33. The abundances of the Belt Sea zooplankton species belonging to different trophic levels during different seasons.

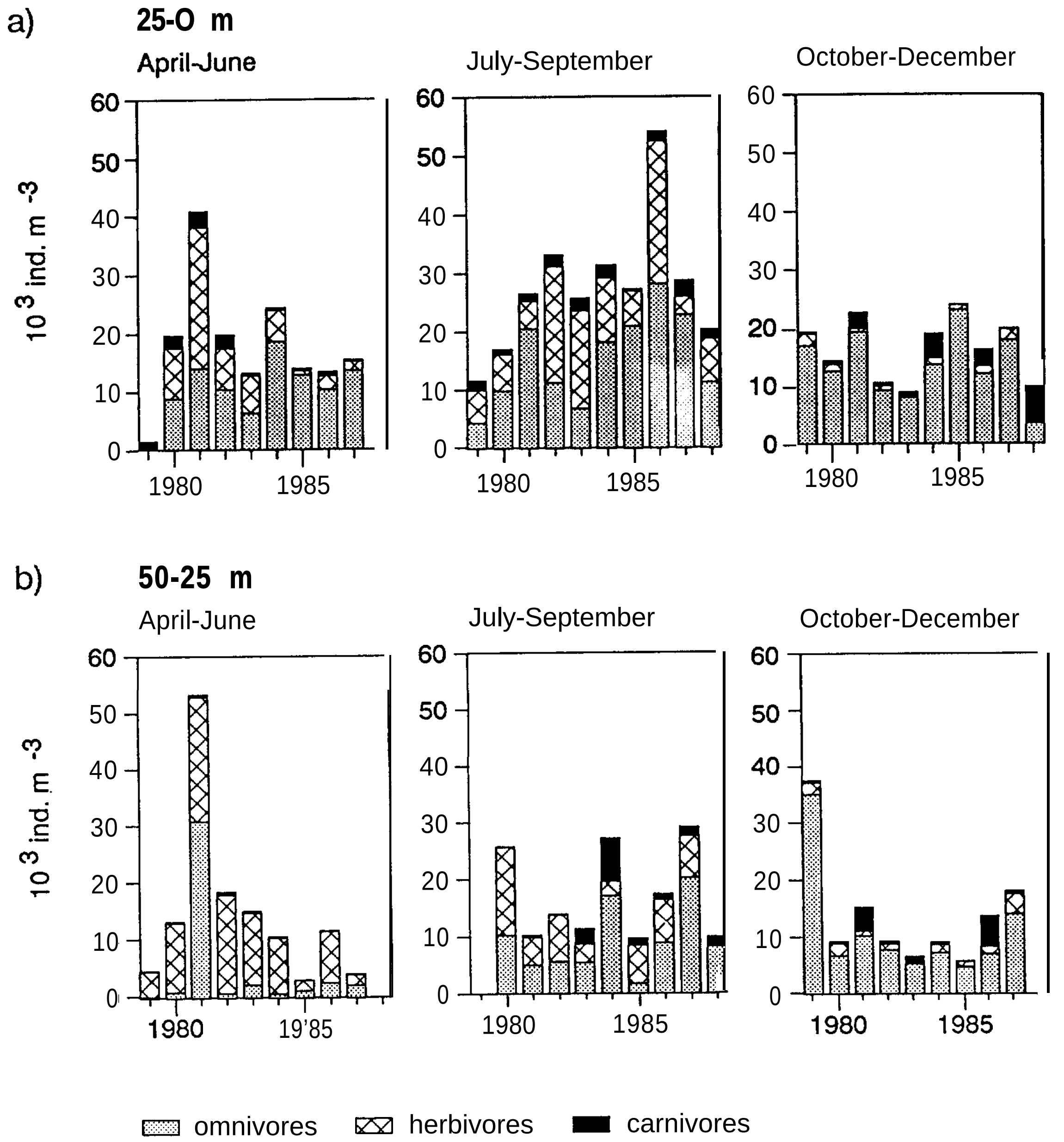

Figure 35. The abundances of the Arkona Sea zooplankton species belonging to different trophic levels during different seasons.

a. 0-20 m depth range

b. 25-50 m depth range

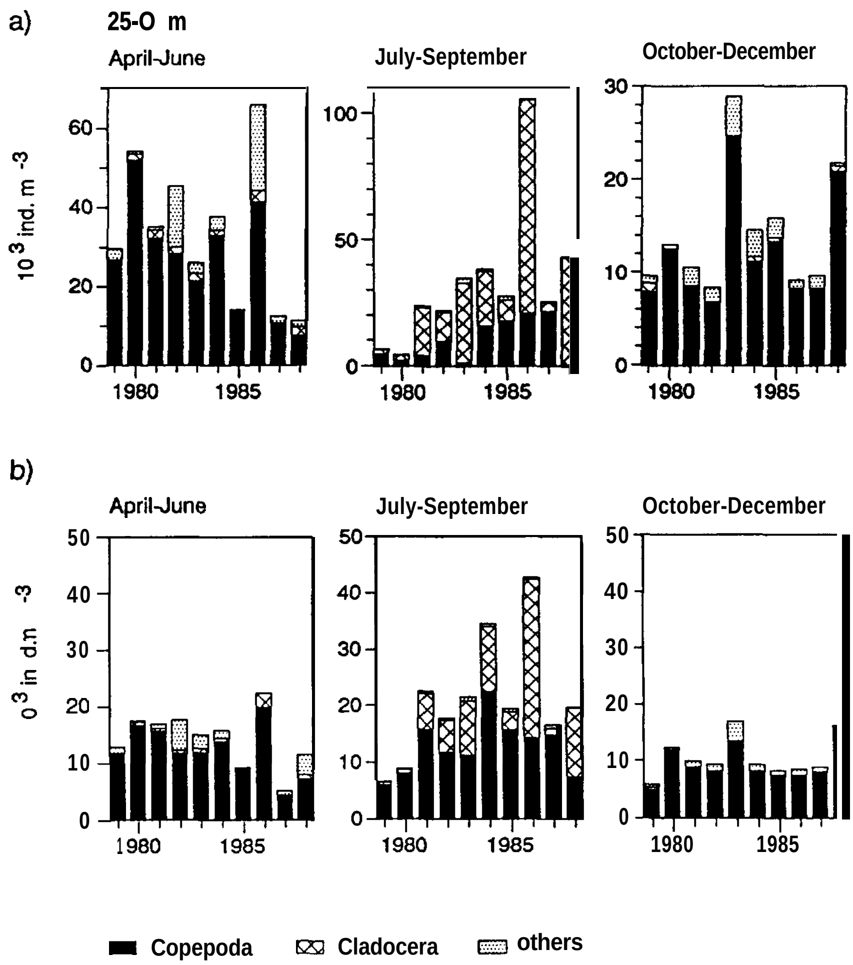

Figure 36. The abundances of the main taxonomic groups in zooplankton in the Bornholm Sea during different seasons.
a) 25 - 0 m,
b) all depths pooled.

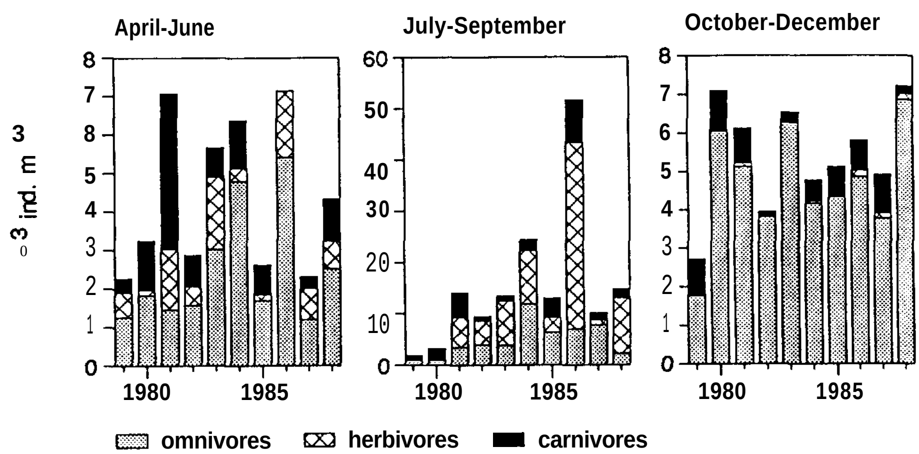

Figure 37. The abundances of zooplankton species belonging to different trophic levels in the Bornholm Sea in different seasons (all depths pooled).

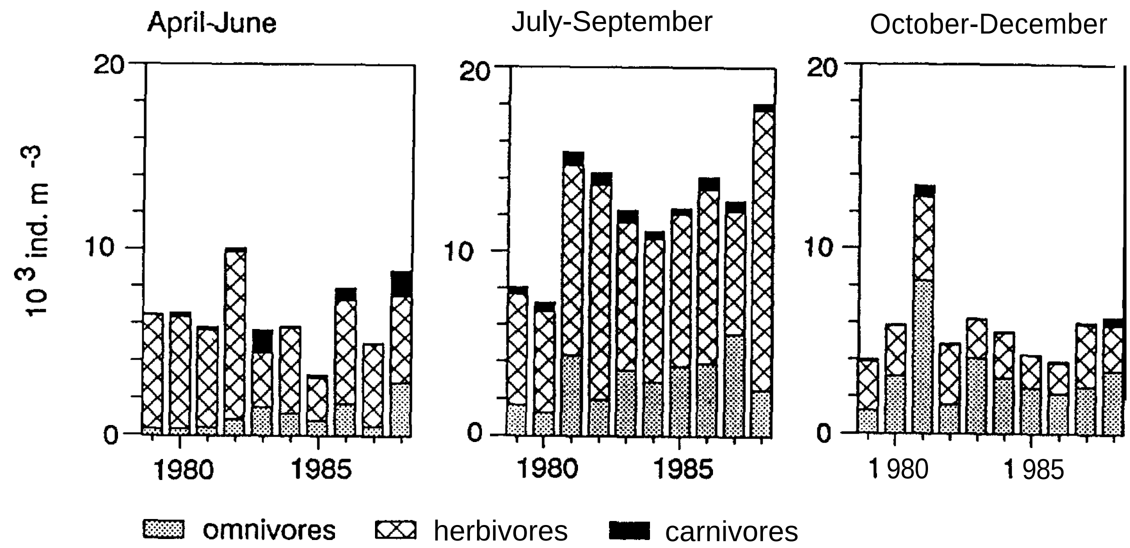

Figure 40. The abundances of zooplankton species belonging to different trophic levels in the Eastern Gotland Sea in different seasons (all depths pooled).

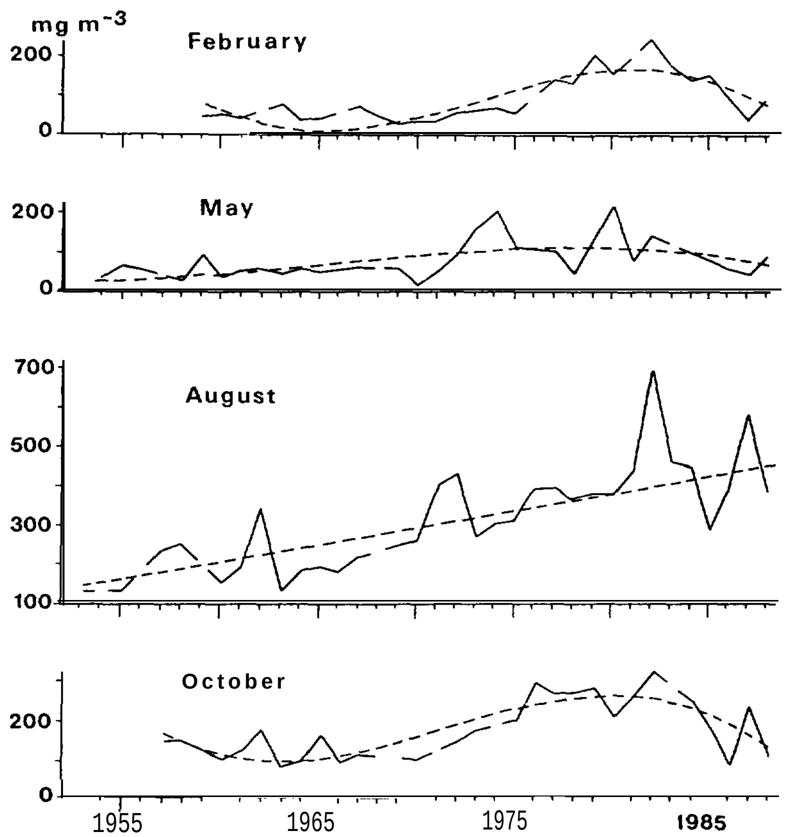

Figure 41. Long-term dynamics of crustacean plankton biomass in the eastern and south-eastern parts of the Baltic Proper in February, May, August and October in 1953-1988 (0-100 m).

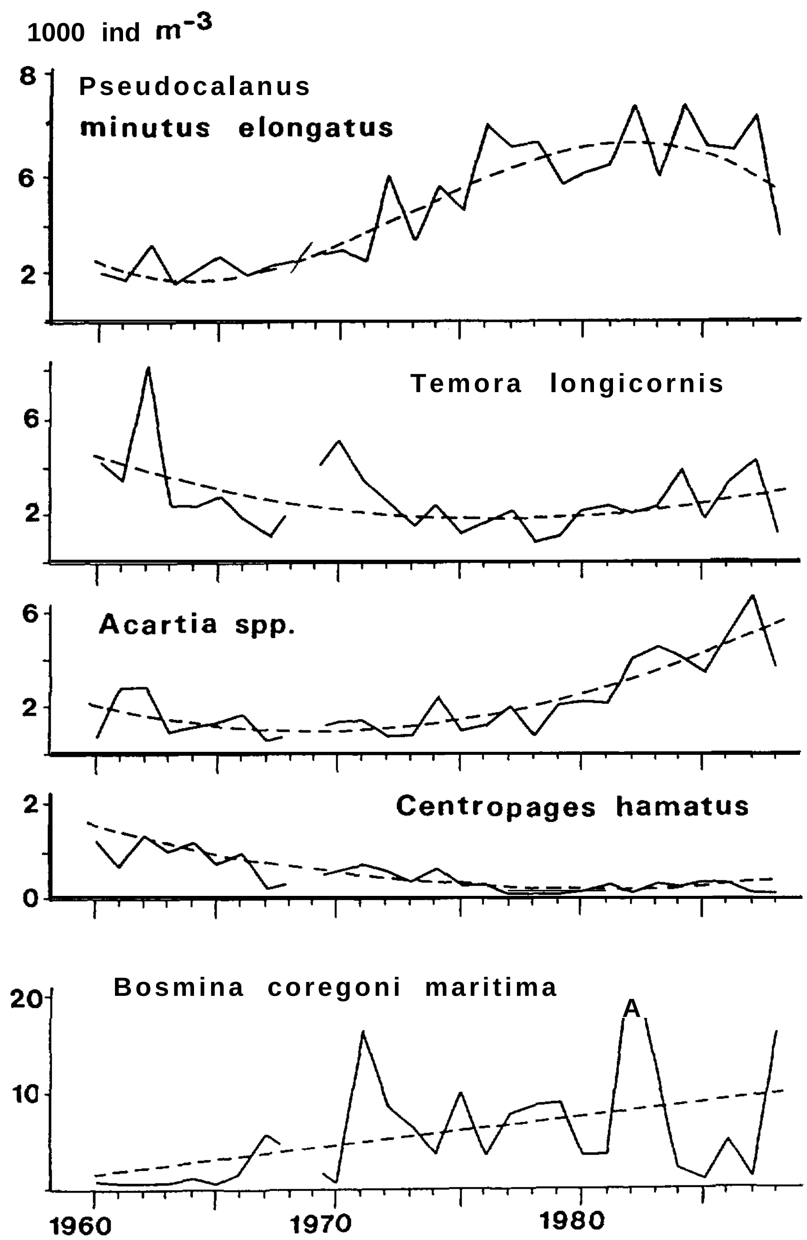

Figure 42. Long-term dynamics of main mesozooplankton species abundance in the eastern and south-eastern parts of the Baltic Proper in August 1960-1988, (0-100 m).

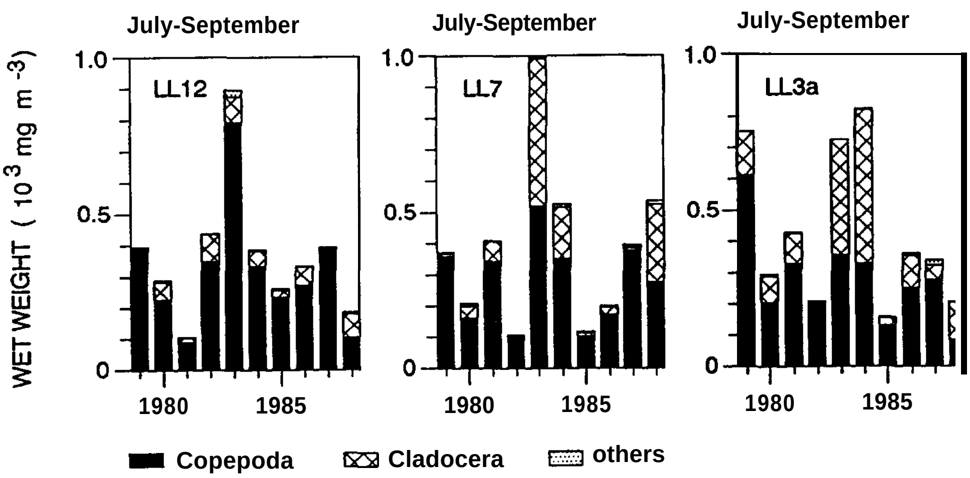

Figure 44. Total zooplankton biomass divided into main taxonomic groups in the Gulf of Finland in summer (July-September) at three stations (LL 12 = BMP H1, LL 7 = BMP F3, LL 3a = BMP F1).

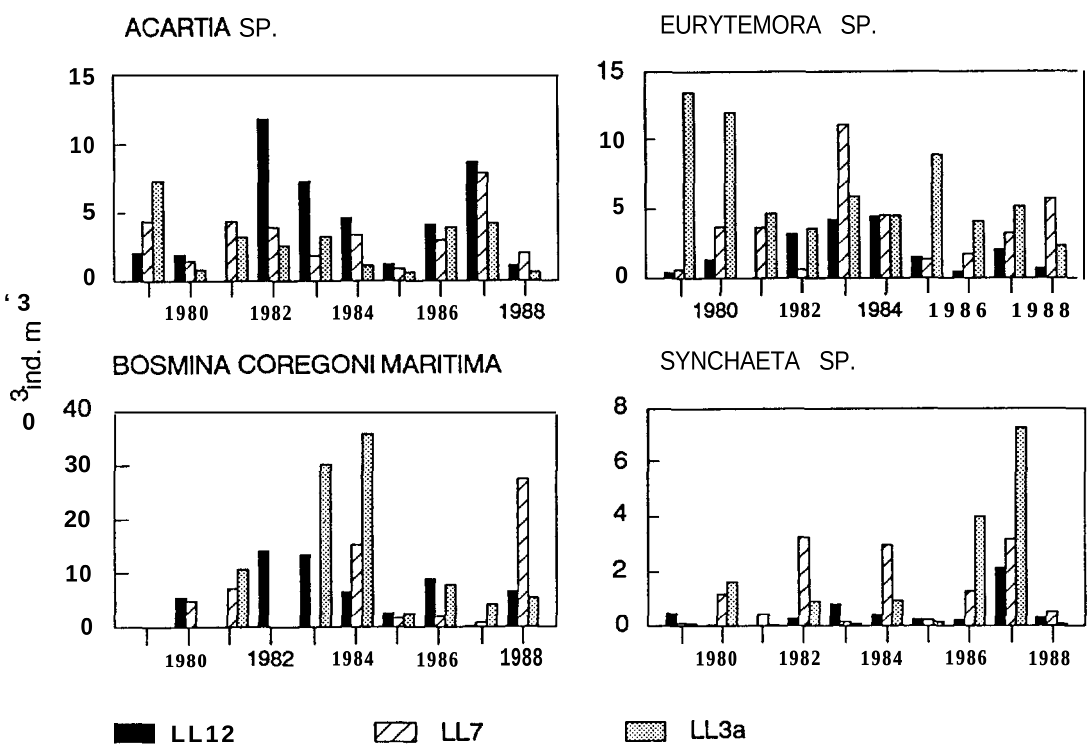

Figure 45. Abundance of the most important species in the Gulf of Finland in summer (July-September) at three stations (LL 12 = BMP H1, LL 7 = BMP F3, LL 3a = BMP F1).

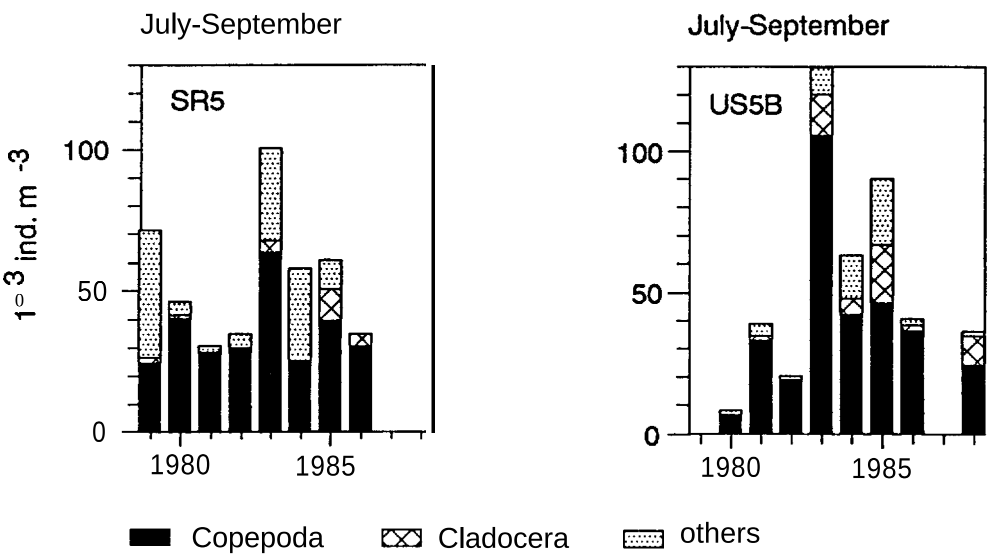

Figure 47. Total zooplankton abundance showing the main taxonomic groups at two stations (SR 5 = BMP C4; US 5b = BMP Cl) in the Bothnian Sea in summer (July-September).

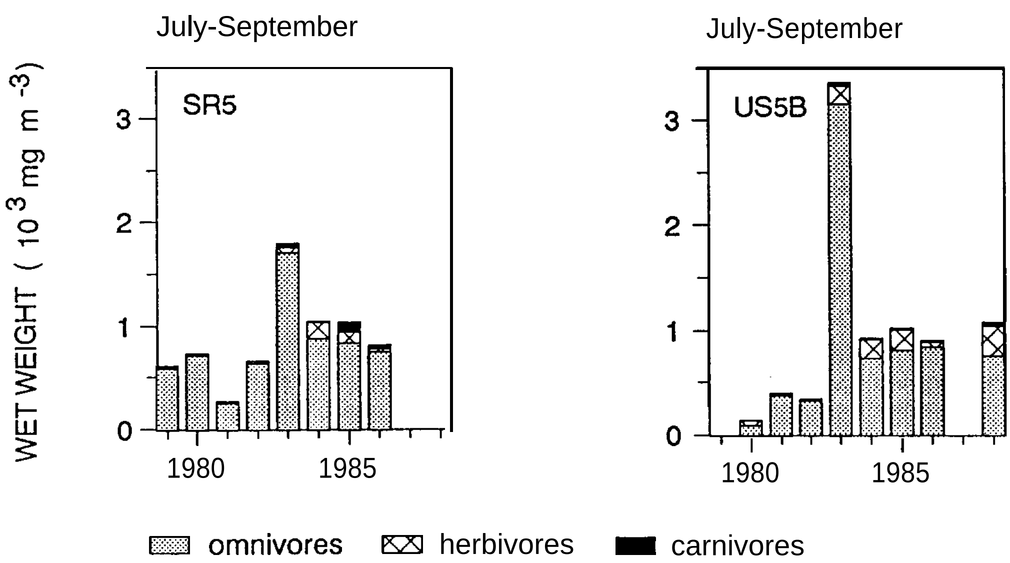

Figure 48. Trophic level of the most important zooplankton species at two stations (SR 5 = BMP C4; US 5b = BMP Cl) in the Bothnian Sea in summer (July-September).

4.7 ASSESSMENT AND REVIEW

S. Schulz[1], G. Behrends[3], G. Breuel[1], U. Horstmann[3], K. Kononen[2], J.-M. Leppänen[2], M. Viitasalo[2], T. Willén[4]

According to the present data further changes in the pelagic ecosystem of the Baltic Sea have been observed in comparison to the First Periodic Assessment (Baltic Marine Environment Protection Commission, 1987).

All the time series about plankton variables reveal a high degree of temporal variability. Some of the interannual fluctuations can be related to the abiotic environmental conditions. Especially the irradiation, the water circulation, and the available nutrient reserves play an important role.

The water above the permanent halocline was affected by the cold winters 1984/1985, 1985/1986, and 198611987. On the other hand the winters 1983/1984 and 1987/1988 were extraordinary warm. The extremes could have affected the duration and extent of convection in spring and thereby the onset of the spring phytoplankton bloom. The development of the zooplankton is also believed to be influenced by the extreme water temperatures.

The irradiation budgets were according to Launiainen (see Chapter "Hydrography") very variable for the years 1984-1988. Only the years 1985 and 1988 displayed values above the long-term mean. The other years were characterised by low irradiation especially during summer in most regions of the Baltic Sea. The high potential primary production observed in some areas in May 1985 and summer 1988 coincide with the maxima in irradiation. In 1986 and 1987 the irradiation was below the long-term mean, and also the primary production was low.

Phosphate and nitrate did not show any longer the increasing trend as described in the First Periodical Assessment. The only exceptions are the Belt Sea and the deep-water in the Baltic basins, where the increasing tendency still holds. The continued reduction in the salinity has resulted in the weakening of the vertical density gradients. The increase in nutrient concentrations of the deep water together with the higher potential for water exchange enhance the possibility of nutrient import to the euphotic layer. There has been an increase in the winter nutrient concentration from 1970s to 1980s. Since the spring phytoplankton bloom consumes all the nitrogen from the euphotic layer, the primary production during the spring growth period must have increased. However, as showed by e.g. the joint patchiness study in the central Baltic Proper (Dybern and Hansen, 1989) the high temporal and spatial variability at that time makes the long-time trend analysis difficult if the sampling frequency is as low as in the present monitoring programme. As shown by Schulz et al. (1985) the amount of nutrients available during the bloom determines not only the absolute values of the phytoplankton parameters but also the duration of the bloom. About half of the spring phytoplankton production sinks out and affects directly the benthic system (e.g. Smetacek et al. 1984, Leppänen 1988). The remaining part is retained in the pelagic system and might influence the pelagic community in summer.

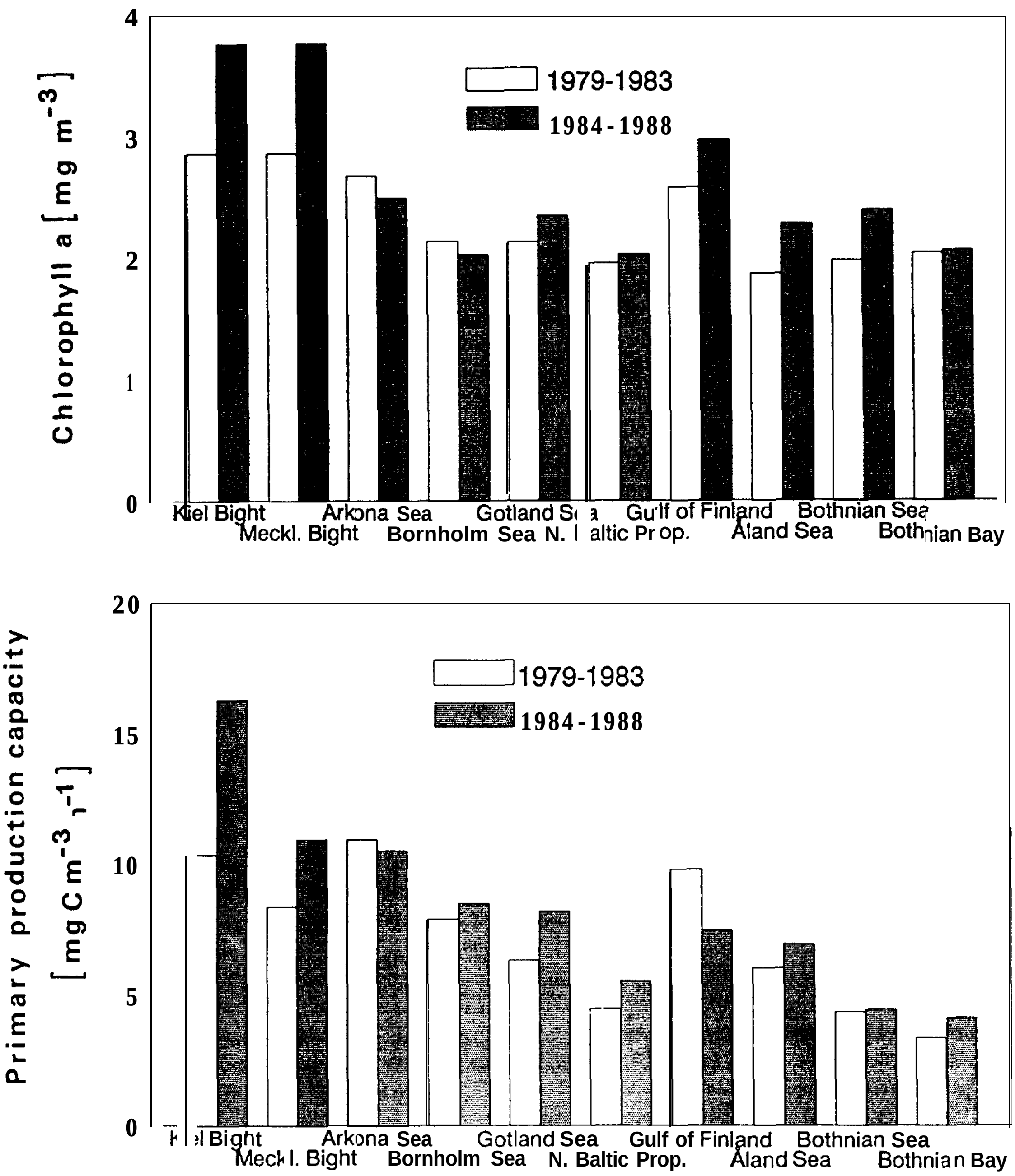

Figure 51. Mean levels of chlorophyll-a (mg m^{-3}) and primary production capacity (mgC m h^{-1}) in the 0-10 m layer of the various areas of the Baltic Sea in 1979-1983 and 1984-1988.

phytoplankton, chlorophyll and primary production in Kattegat without finding any marked changes during the period 1979-1981 *or* compared with results gained during the 1970's (Björn-Rasmussen 1976, Edler 1977, Thomsen 1979). Other background data from the beginning of this century are given in a number of publications by Cleve (1902). Pihl Baden (1986) discusses both phytoplankton and zooplankton in Skagerrak and Kattegat and concludes: "In both the Kattegat and the Skagerrak, prolonged spring blooms, especially of dinoflagellates, have been registered. Furthermore, in the Kattegat, dinoflagellates show a tendency to increase in number of species among which some are potentially poisonous".

It has been discussed if the number of flagellate species in the Baltic Sea has increased during the last years. Rybicka et al. (1990) showed in three stations in the Bornholm and Southern Gotland Sea since 1983 a remarkable appearance of non-identified flagellates. It is uncertain if this is an artifact that the flagellates have been overlooked in the samples before or this is a new feature of the ecosystem. In the latter case this could be treated as a sign for eutrophication. According to the present data no general trend, however, is discernible supporting such an assumption except for the Kattegat-Belt Sea region.

In May/June 1988 an exceptionally strong bloom of ***Chrysochromulina polylepis*** has developed in the Skagerrak-Kattegat area. In connection with the mass appearance of the small flagellate, kills of benthic organisms (macroalgae, molluscs, polychaets, sea stars) and caged fish, were reported. Wild fish (cod and others) were also infected but could in most cases escape from the bloom regions (Horstmann and Jochem 1988, Dahl et al., in press., Skjoldal and Dundas 1989).

The water inflow/outflow of the Baltic Sea highly influences the phytoplankton (Kell 1984). The phytoplankton response on the increasing nutrient concentrations indicates generally that the highest total biomass were reached in the period 1979-1981, followed by some low-biomass years and again increasing values from 1985 on. The development is, of course, variable in different sea areas. Increasing diatom biomass were observed from 1985 in several areas, e.g., Bornholm Sea, Eastern and Western Gotland Sea and Gulf of Finland resulting in the decrease of silicate pool in the euphotic layer. Decreasing silicate values then may lead to a change in the diatom species composition, an alteration worth to observe.

Statistically significant (Mann-Whitney test, probability level of 0.1) biomass increase at the reference stations BO 3, SR 5, F 64 and BY 5 in spring, at stations F 64 and LL 7 in summer, and at stations BO 3, F 64, LL 7, BCS III 10 and Anholt in autumn was found when compared the first (1979-1984) and the second (1985-1988) investigated period. A decrease in the summer biomass values at BY 15 and Halsskov was observed. These data, however, give no general tendency to the Baltic as a whole. This is perhaps not to be expected due to the variation in different subsystems with varying reactions to differing loading. Another main reason is the short investigation period with too sparse samplings.

Regarding the increase in the primary production, one could expect an increase in mesozooplankton, too. However, positive trends of the abundance could be found only in summer in the Arkona Sea and in the Eastern Gotland Sea and the tendency of increase in the Bornholm Sea. For in the Baltic Proper and in the Gulf of Bothnia. The sources of the eutrophication are related to nutrient input of anthropogenic origin and remobilized reserves from the deep water.

SUMMARY

Signs of eutrophication are evident in the pelagic ecosystem of the Baltic Sea although no drastic changes either on the quantitative level nor in the species composition have occurred in the open pelagic system since the First Periodic Assessment.

When comparing the assessment periods 1979-1983 and 1984-1988, primary production showed increasing tendency in most areas. In the Arkona Sea and in the Gulf of Finland the summer primary production was decreasing while in the Bothnian Sea it stayed at the same level. Chlorophyll a values showed an increasing tendency in most areas, except in the Arkona Sea and in the Bornholm Sea.

In general, no clear changes in the phytoplankton species composition have been detected. Several exceptional and in some cases toxic phytoplankton blooms have been reported. Obviously the blooms of dinoflagellates, chrysophyceans and prymnesiophyceans have been more frequent in the Kattegat-Belt Sea region than in the Baltic Proper, where in most cases the blooms were caused by cyanobacteria. Several potentially toxin producing species of cyanobacteria, dinoflagellates, chrysophyceans and diatoms occur in the Baltic Sea and are a risk for noxious blooms. In the Kattegat-Belt Sea region, an increase of flagellates was observed.

The zooplankton abundance values in 1979-1988 revealed large interannual fluctuations. Increasing trends could be detected in the Arkona Sea and in the Gotland Sea in summer, a tendency of increase in the Bornholm Sea.

The assessment period 1979-1988 is still short for proper trend analysis. It has revealed large interannual variations in the plankton communities which can partly be related to large climatic fluctuations. The plankton community is controlled by various factors which result in different temporal and spatial scales for the response to the environmental changes. These variations can largely hamper the effects of the e.g. still large input of nutrients. The anthropogenic disturbances increase the variability of the system as well. The few long time series available on plankton reveal clear signs of eutrophication. During the last 30 years the phytoplankton primary production has almost doubled from the Kattegat to the Baltic Sea proper. The phytoplankton biomass expressed as chlorophyll a increased nearly by the same order of magnitude. Due to the development in methods and to the increasing knowledge of systematics, long comparable time series on phytoplankton is not available. The phytoplankton composition has not changed drastically but there is a tendency to more frequent, and sometimes toxic blooms. Long-term investigations in the Bornholm and Gotland Sea show a significant increase in the zooplankton abundance from 1950s to the end of 1980s with the steepest increase in the 1970s.

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5. ZOOBENTHOS

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ABSTRACT

The macrozoobenthos part of the assessment of the state of the Baltic Sea is based mainly on the observations made within the Baltic Monitoring Programme, coordinated by the Helsinki Commission. The data from national monitoring programmes have been used to evaluate the state of nearshore areas.

In the entrance area to the Baltic Sea, increased input of organic material to the bottoms have led to an increase *in* zoobenthos biomass in the northern part of the Kattegat. In the southern part of the Kattegat just below the haldcline, in the deeper areas of the Belt Sea and the Arkona Sea the reduction in zoobenthos biomass and even macrofauna death is coupled to a more frequent occurrence of seasonal oxygen deficiency.

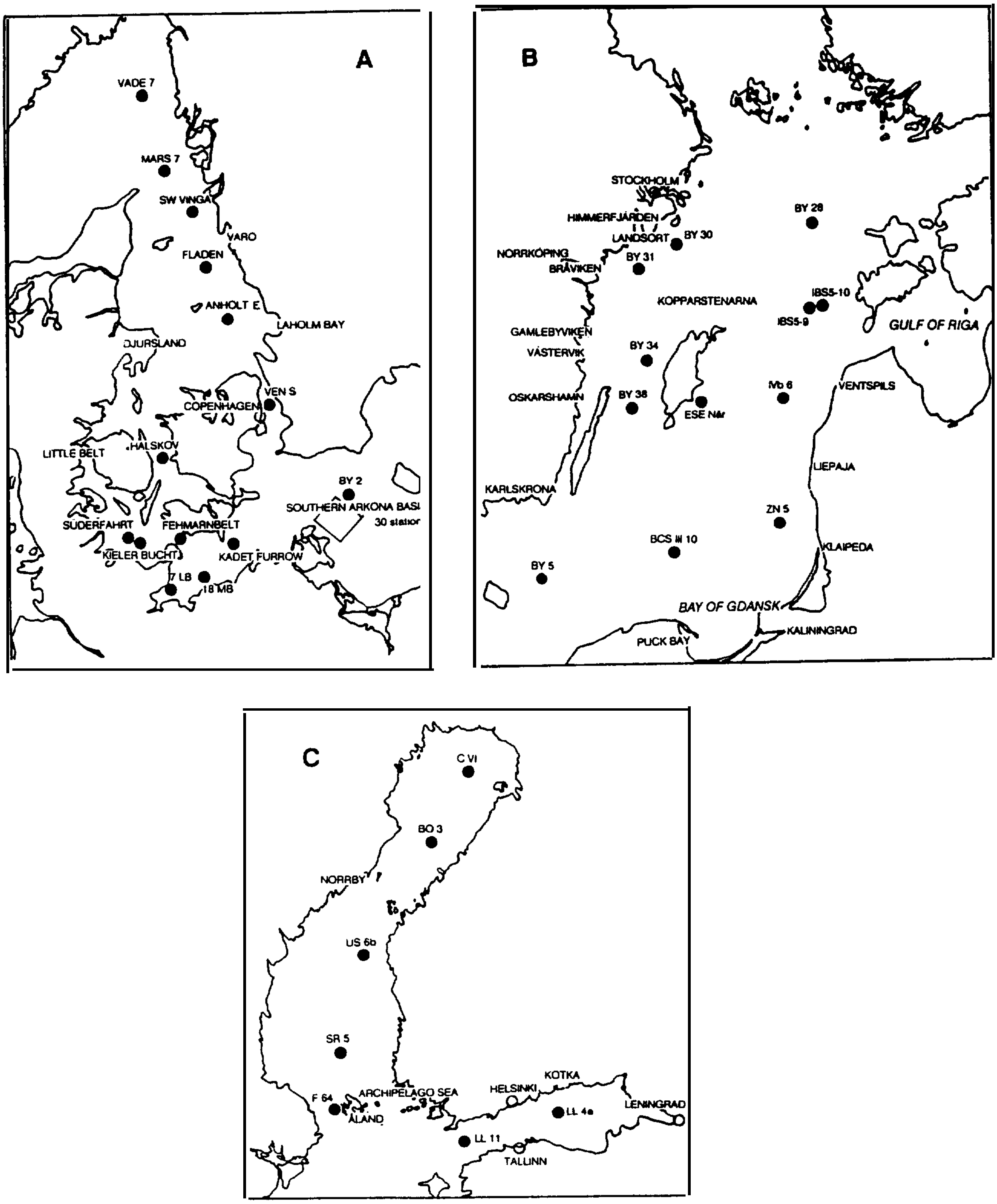


Figure 1. Macrozoobenthos investigation localities in the Baltic Sea. A – Kattegat, the Sound, the Belts, Belt Sea and Arkona Basin, B – Baltic Proper (for localities in the south-eastern Baltic Proper, see Fig. 20), C – Gulf of Finland and Gulf of Bothnia.

Kiel Bay

Most of the image material is taken at various national stations in the Kiel Bay. Nevertheless, in this context only material from the station Siiderfahrt (BMP N2) is presented, that shows a soft muddy sediment which turns from brown into black deeper than 4-5 centimetres. At the national station Boknis Eck in the western part of Kiel Bay every year we experience phases with oxygen depletion and the formation of H_2S in the water. This is reflected by an accumulation of black flocculent material floating over the sediment surface and only visible by imaging methods. This phenomenon can be found in all Baltic basins investigated so far and will be dealt with below.

Arkona Basin

During a monitoring cruise with RV "Littorina" in June 1989 TV-recordings revealed wide-spread spots of sulfur bacteria *(Beggiatoa)* on the sea floor surface together with dying macrofauna. These white spots seem to grow in small circles (10-20 cm) around sources of organic material, which in some cases come from dead *Arctica* or from organic material sedimented in shallow depressions in the sea floor. The oxygen values of the overlying water were, nevertheless, not an indicator for such a severe shift (50 % saturation). Grab samples as well, as dredge tows showed no living macrofauna at BY 1 (BMP K7), a station that in the preceding years - although impoverished - was still vital. Additional sediment-profile photographs (BEMOTS) showed a black organic layer on top of the normal muddy sediment covered by the above mentioned layer of sulfur bacteria. In 1988 the image profiles had been without these detrimental features.

As seen from other Baltic basins these findings indicate a severe change in the ecosystem. Before 1989 there was a rich mollusc-dominated bottom fauna community which served as good food for bottom living fish. Food items for fish are now very scarce on these bottoms. The severe consequences for the fish populations can only be speculated on at this moment.

Bornholm Basin

Video inspections from earlier years showed a layer of flocculent material in the zone just above the sediment surface, which was drifting with the currents *over* the bottom. The size of these flakes increased generally from West to East, starting in the Arkona Basin. As mentioned above this aspect have been found also in locally restricted areas in the Kiel Bay In the northern part of the Bornholm Basin (below 75 m) these flakes are condensed by *Beggiatoa* lawns and can no longer move over the bottom. It seems that the northern part of the Bornholm Deep is more deteriorated than the southern part.

5.2.2 Conclusions

The evaluation of video profiles and photographs both from the surface and vertical profiles (REMOTS) gives us information of the state of the sediment habitat that will not be gained with traditional approaches. Although results are scattered some general features can be derived from this source of information. The crucial point in all Baltic basins seems to be the presence/absence of macrofauna to re-work the sediment and to take part in the remineralization of the detritus coming from the pelagial and by lateral advection. The shift from one state to the other is characterized by dense layers of detritus flakes at the sediment surface and in the overlying water. The size of the flakes is increasing from West to East in the Southern Baltic Proper which is interpreted as an increasing deterioration. After the oxygen in the sediment is depleted these flocculent layers are consolidated by *Beggiatoa* mats which can only exist at the boundary layer between H_2S-mud and water with at least some oxygen. This was the case in the northern part of the Bornholm Deep, the Gdansk Deep, the southern part of the Eastern Gotland Basin and the Landsort Deep (Northern Baltic Proper). One can also follow up the genesis of laminations in this geographical order. The beginning of this detrimental status could be observed temporally and locally restricted in the Arkona Sea and in parts of the Kiel Bay. There are indications, that some parts of the southern Kattegat are also close to this severe alterations.

The basins north of the Åland Sill show a completely different picture with fine sediments strongly re-worked by the activity of macrofauna and with no signs of reduction mediated by oxygen deficiency.

5.3 REGIONAL ASSESSMENTS

5.3.1 The Kattegat, the Sound and the Great Belt

J. N. Jensen[4] and A. Josefson[4]

Kattegat, the Sound and the belts are rather shallow sea areas. Water deeper than 25 m is confined to the eastern side of Kattegat and the central part of the Sound and the belts where the sediment generally consists of silt and clay. The shallower areas on the western side have in general coarser sediments. Water circulation in the Kattegat is complex but tends to be dominated by a general inflow of saline North Sea water at depth from the Skagerrak towards the Belt Sea and the Sound and an outward surface flow of brackish Baltic water in the opposite direction.

The macrobenthic infauna is being monitored once a year at 5 stations below the halocline (Fig. la). The sampling started in 1979 at the two northern stations in the Kattegat, SW Vinga = GF 4 (BMP R7) and Fladen (BMP R6) and at one station in the Sound, S Ven (31 S = BMP Q1). At the southern station in the Kattegat, Anholt E (413 = BMP R3) and the station in the Great Belt, Halskov (939 = BMP P1) sampling was started in 1982. Station Anholt E has been sampled two times in spring by different laboratories, one in Denmark and one in Sweden.

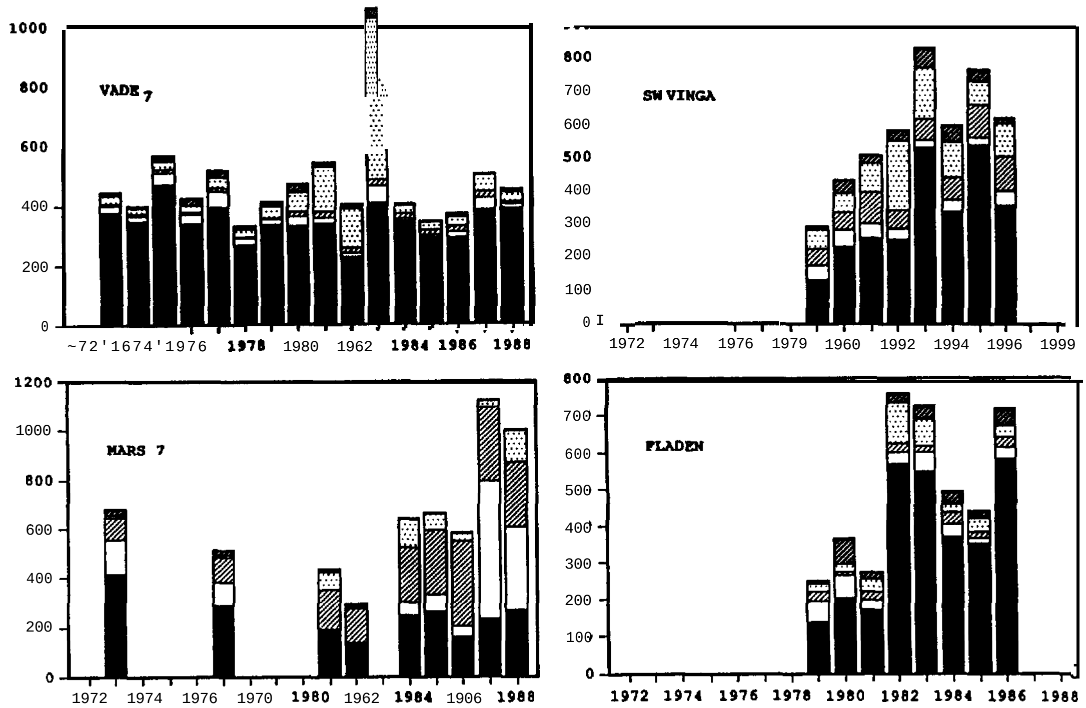


Figure 2. Cumulative histograms showing the composition of abundance (ind./0.1 m^2) at 4 stations in the Skagerrak and northern Kattegat from the period 1973-1988. The areas in the histograms denote frombottomtotop: Polychaeta, Mollusca, Echinodermata, Crustacea and Miscellaneous.

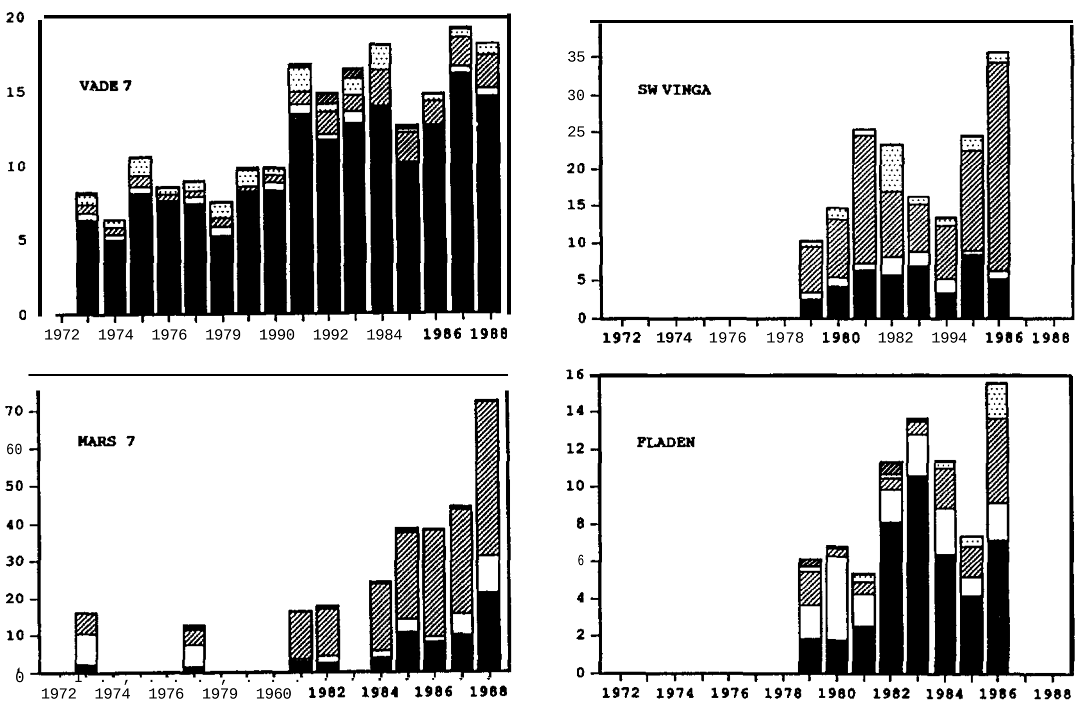


Figure 3. Cumulative histograms showing the composition of biomass (mg/0.1 m^2) when echinoids and *Arctica* have been excluded, at 4 stations in the Skagerrak and northern Kattegat from the period 1973-1988. For further information see previous figure.

Table 2. Results of linear regression analysis on biomasses versus year at 7 stations in the Skagerrak-Kattegat area from the time period 1973-1988. For further information see Table 1.

Station	Depth (m)	Period	n-1	Slope	P	Sign
Total biomass						
VADE7	100	1981-88	39	+	0.047	*
MARS7	100	1981-88	34	+	0.000	***
SW VINGA	75	1979-86	23	+	0.018	*
FLADEN	75	1979-86	23	+	0.180	ns
ANHOLT E	55	1982-88	6	-	0.658	ns
S VEN	17	1982-88	6	+	0.830	ns
HALSKOV	28	1982-88	6		0.089	ns
Biomass without large species						
VADE7	100	1973-88	79	+	0,000	***
MARS7	100	1973-88	42	+	0.000	***
SW VINGA	75	1979-86	23	+	0.003	**
FLADEN	75	1979-86	23	+	0.014	*
ANHOLT E	55	1982-88	6		0.216	ns
S VEN	17	1982-88	6	+	0.830	ns
HALSKOV	28	1982-88	6		0.018	*
Polychaeta						
VADE7	100	1973-88	79	+	0.000	***
MARS7	100	1973-88	42	+	0,000	***
SW VINGA	75	1979-86	23	+	0.248	ns
FLADEN	75	1979-86	23	+	0.014	*
ANHOLT E	55	1982-88	6	+	0.352	ns
S VEN	17	1982-88	6	+	0.534	ns
HALSKOV	28	1982-88	6		0.140	ns
Mollusca						
VADE7	100	1973-88	78		0.652	ns
MARS7	100	1973-88	42	-	0.650	ns
SW VINGA	75	1979-86	23	-	0.996	ns
FLADEN	75	1979-86	23		0.428	ns
ANHOLT E	55	1982-88	6	+	0.596	ns
S VEN	17	1982-88	6	-	0.593	ns
HALSKOV	28	1982-88	6	+	0.535	ns
Echinodermata						
VADE7	100	1981-88	39	+	0.024	*
MARS7	100	1981-88	34	+	0.007	**
SW VINGA	75	1979-86	23	+	0.028	*
FLADEN	75	1979-86	23	+	0.321	ns
ANHOLT E	55	1982-88	6	-	0.396	ns
S VEN	17	1982-88	6	+	0.570	ns
HALSKOV	28	x982-68	6	-	0.091	ns
Amphiura spp.						
VADE7	100	1973-88	79	+	0.000	***
MARS7	100	1973-88	42	+	0.000	***
SW VINGA	75	1979-86	23	+	0.011	*
FLADEN	75	1979-86	23	+	0.011	*
ANHOLT E	55	1982-88	6	-	0.030	*
S VEN	17	1982-88	6	+	0.590	ns
HALSKOV	28	1982-88	6		0.482	ns
Crustacea						
VADE7	100	1973-88	79		0.791	ns
MARS 7	100	1973-88	42	+	0.043	*
SW VINGA	75	1979-86	23	+	0.966	ns
FLADEN	75	1979-86	23	+	0.552	ns
ANHOLT E	55	1982-88	6	-	0.033	*
S VEN	17	1982-88	6	-	0.138	ns
HALSXOV	28	1982-88	6		0.080	ns
Miscellaneous						
VADE7	100	1973-88	79	+	0.292	ns
MARS7	100	1973-88	42	-	0.931	ns
SW VINGA	75	1979-86	23	+	0.290	ns
FLADEN	75	1979-86	23		0.159	ns
ANHOLT E	55	1982-88	6	-	0.370	ns
S VEN	17	1982-88	6		0.410	ns
HALSKOV	28	1982-88	6	+	0.117	ns

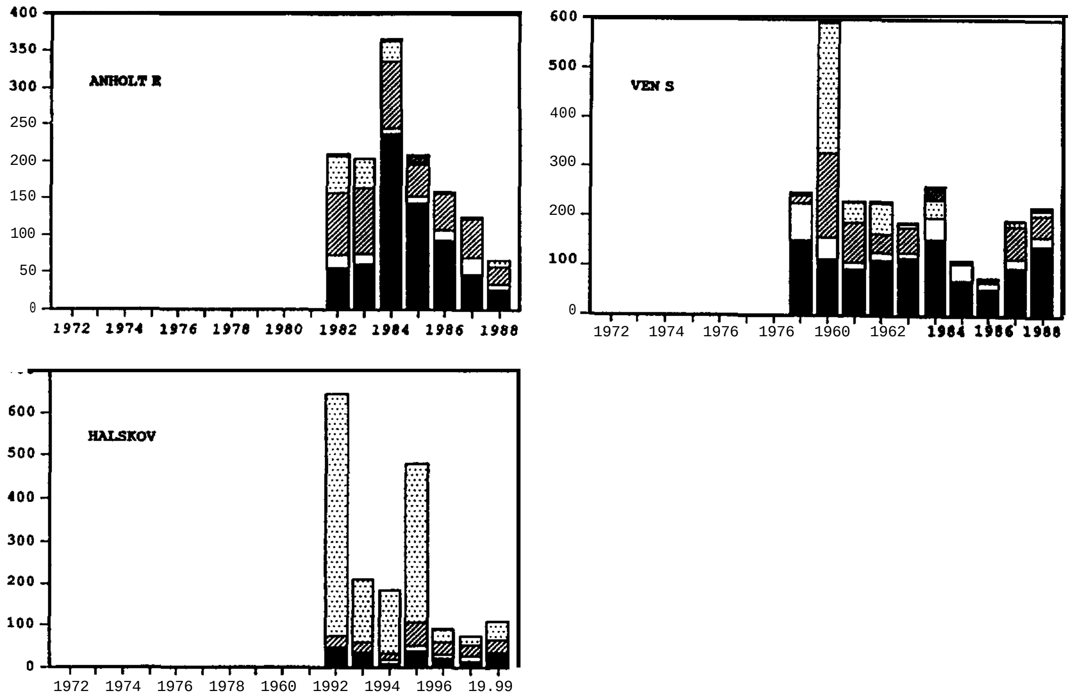


Figure 4. Cumulative histograms showing the composition of abundance (ind./O.l m^2) at 3 stations in southern Kattegat and the Sounds from the period 1979-1988. For further information see Figure 1.

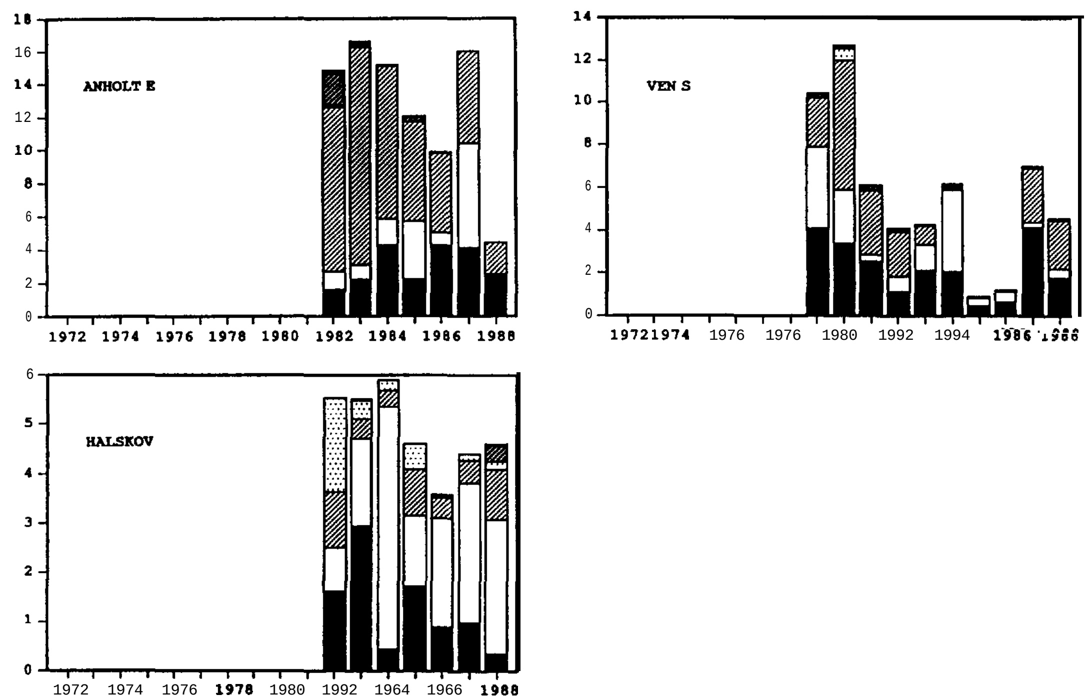


Figure 5. Cumulative histograms showing the composition of biomass (mg/O.1 m^2) when echinoids, Psolus, Neptunea and Arctica are excluded, at 3 stations in southern Kattegat and the sounds. For further information see Figure 1.

Conclusions

The increasing trend in biomass in the northern part of the Kattegat and the Skagerrak as well is most easily explained by an increased input of organic material to the bottom. This explanation is supported by the fact that the increase involve8 several different species from different trophic groups and with different life spans. Furthermore the change is due to not only increase in abundance but also in individual size. At the northern stations there are no evidence so far of oxygen levels that could have a limiting effect on the macrobenthos.

The indifference or in some cases reductions in abundance and biomass observed in the southern part of the Kattegat, the Sound and the Great Belt towards the end of the investigation period may be related to the hypoxic conditions which appear to have increased in area1 extent in recent years. Oxygen levels in the bottom water show a clear decreasing trend in the 19808 at all stations in the Kattegat and the lowest levels, less than 2 ml/l, were reached at the southern stations. However, these measurements does not give a clear clue as to whether hypoxia really has been the limiting factor. For instance the years with lowest oxygen levels, such as 1986 at Anholt E, was not followed by significant reductions the year after. Furthermore even the lowest levels reached are, according to the literature, above the limit for mass mortality of adult macrofauna. It is possible, though, that the oxygen measurements, which were made some meter above the bottom, are not representative for the near bottom water which the benthos experience (e.g. Rosenberg & Loo, 1989). Furthermore, even if adult individuals are not affected it is possible that recruitment has been adversely affected by the low oxygen levels.

The situation in Kattegat seems to be a result of a general eutrophication as no general trend in salinity and temperature i.e. water circulation has been observed in this area throughout the sampling period. An alternative to eutrophication is a possible reduction of the stocks of demersal fish-species, and thereby of the predation pressure on the benthos. The increase8 in biomass on the northern stations may be interpreted as a result of lowered predation intensity. However, there are no data to indicate that fish stocks have declined in this area (see Chapter "Baltic Fish"). The reactions of benthos in southern Kattegat can definitely not be explained by this hypothesis.

5.3.2 The Belt Sea

F. Gosselck[3] and H. Rumohr[6]

The southern parts of the Belt Sea are shallow with depths ranging to 28 m. Areas below 20 meter are covered with soft mud, whereas the sediment in shallower regions are increasingly sandy with decreasing water depth (Babenerd & Gerlach 1987). The water is stratified in late summer, and oxygen deficiency occur periodically in some areas. There are no major fresh water inflows, and the input of nutrients is mostly domestic and from agricultural sources. The first quantitative benthos investigations were done by C.G.Joh.Petersen in the early century, followed by Hagmeier 1924-1931 (Rumohr 1987). After World War II Kühlmorgen-Hille investigated the benthos in the Kiel Bay from 1952 to 1965, followed by Arntz in 1968,

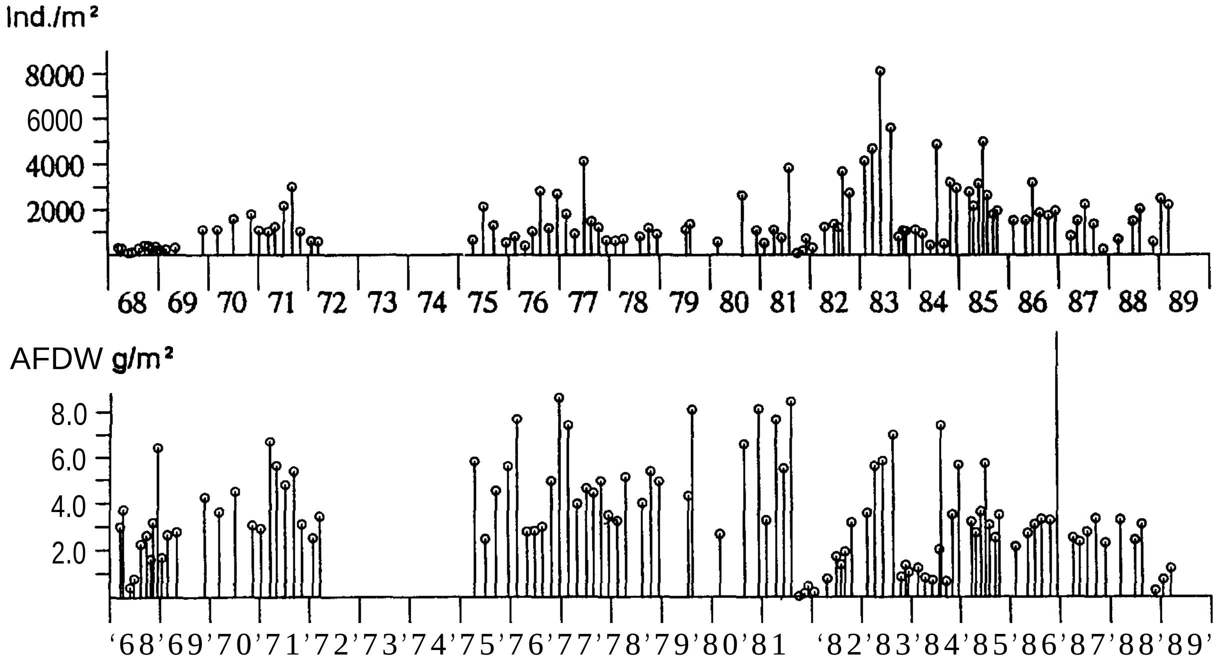


Figure 6. **Total abundance and biomass at station Siiderfahrt (BMP N2) 1968-1989 *(Arctica, Astarte* excluded).**

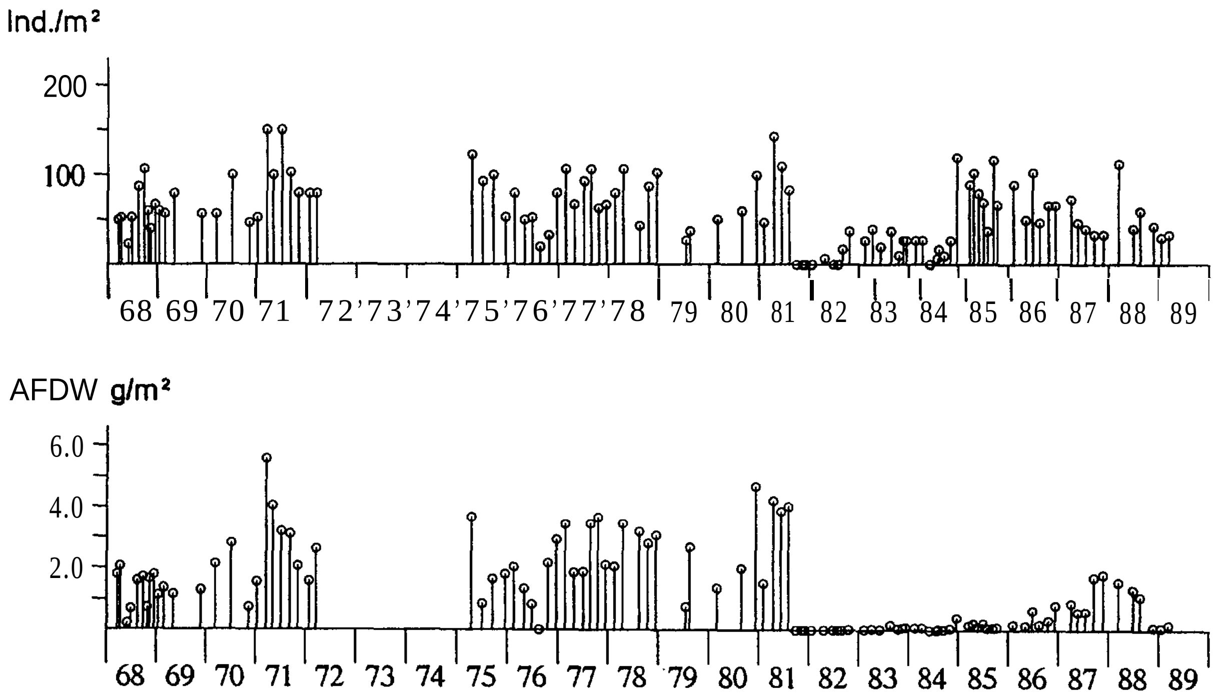


Figure 7. **Abundance and biomass of polychaete *Nephtys* spp. at station Siiderfahrt (BMP *N2)* 1968-1989.**

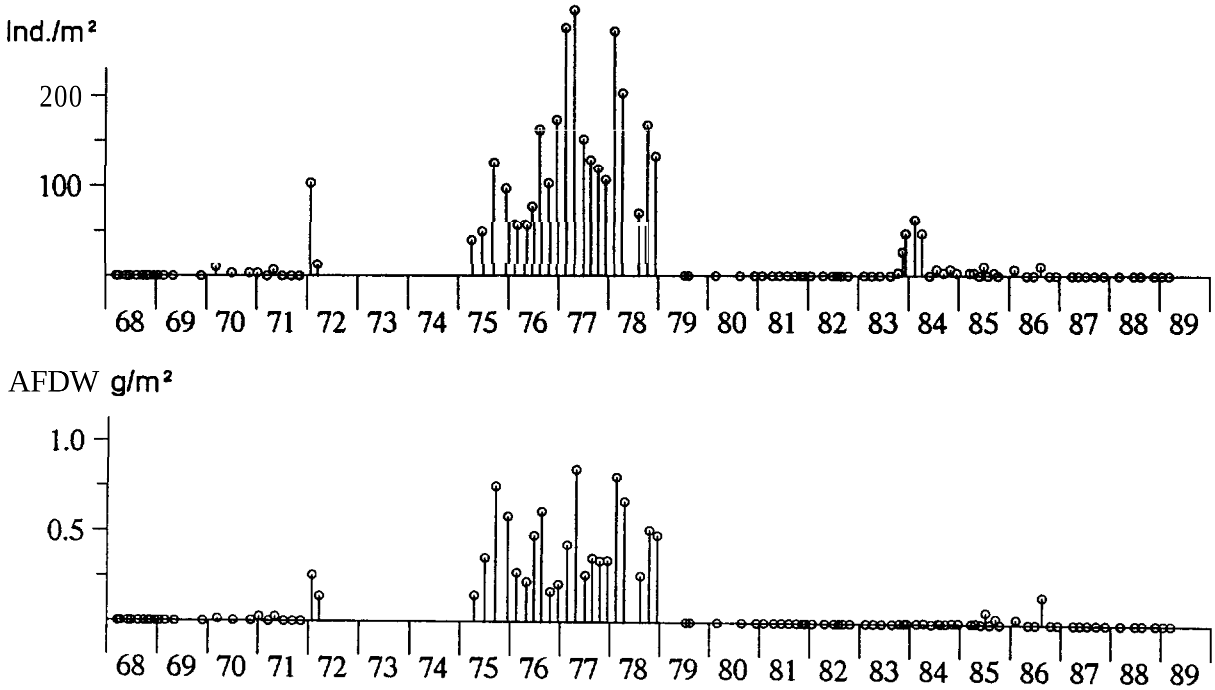


Figure 10. Abundance and biomass of brittle star *Ophiura albida* at station Siiderfahrt (BMP N2) 1968-1989.

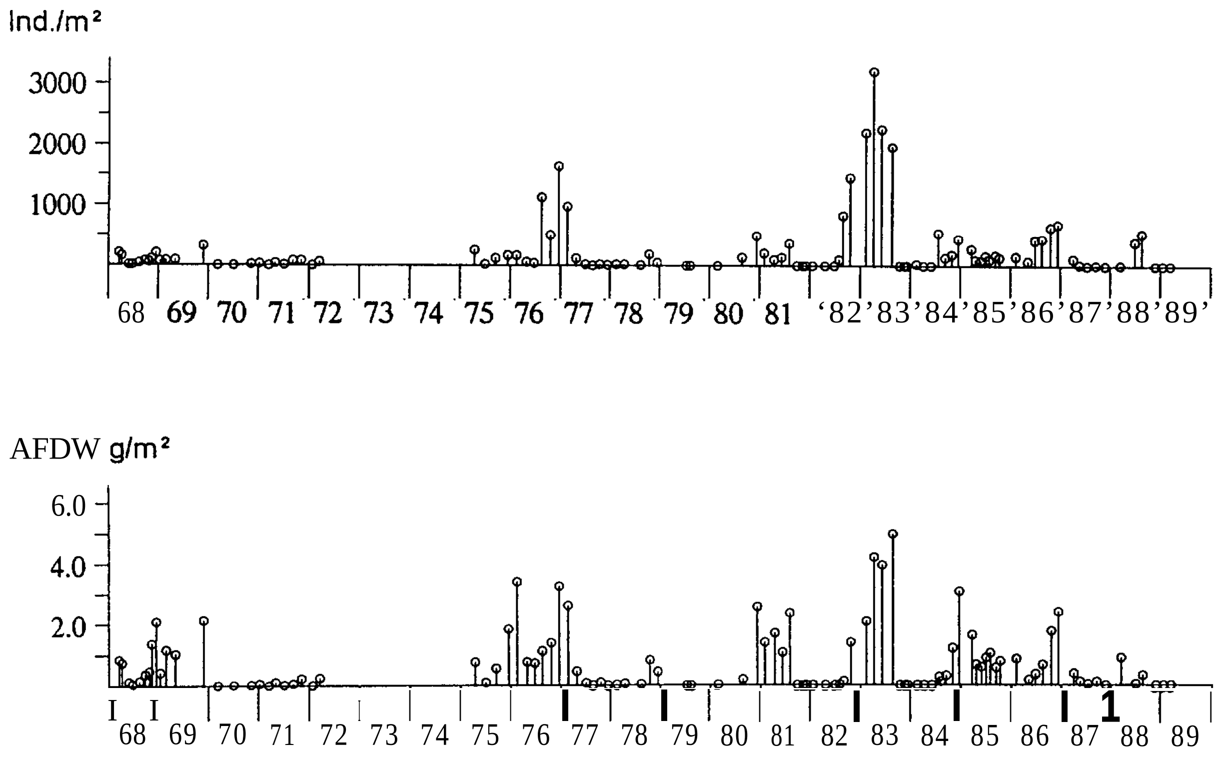


Figure 11. Abundance and biomass of bivalve *Abra alba* at station Süderfahrt (BMP N2) 1968-1989).

The western part of Kiel Bay is affected by oxygen depletions almost every year but the depth range of these events may not every year affect the national in-shore station, Boknis Eck (20m) (Fig.1a) situated on a slope at 20 m depth. Nevertheless, the patterns observed here in general correspond with those of the Süderfahrt station although the fluctuations are wider at this more disturbed station.

The results of the shallow water stations in Kiel Bay (incl. station Kieler Bucht = BMP N3) (Fig. la) show a rich species spectrum dominated by molluscs, that remained stable throughout the last years. All stations reveal a distinct seasonal cycle as have been reported before. There is no major trend in these data, the time covered may, however, also be too short, as has been explained above. Brey (1986) found significantly higher mollusc biomasses in the 1980s compared to the 1960s, in above halocline Kiel Bay sediments.

The station Fehmarnbelt (BMP N1) (Fig. la) was sampled twice a year only since 1986 but the available data from 1981/82 show that the station was affected by the wide ranging oxygen depletion in 1981. Nevertheless there seems to bee a continuous decline in most population parameters (Fig. 13). Only the bivalves (*Abra alba*) had a maximum in 1987/88. This negative process is underlined by the increase of *Halicryptus spinulosus* which seems to be key species for impoverished soft bottoms (Fig. 14). Whether this deterioration is linked to oxygen problems in the southern Kattegat and in the Arkona Basin must be investigated in future.

Conclusions

The available data set covers 20 years and shows that the data from the period 1984-1988 alone are not sufficient, in this specific sea area, to reveal the real changes in an ecosystem, that is governed by strong hydrographic fluctuations in combination with wide ranging oxygen depletions, which has been becoming a regular phenomenon in the autumn of almost every year since 1981. The fauna1 successions following major disturbances like the ice winter in 1978/79 and the oxygen depletion in 1981 and 1983 *govern* the image of the benthic community over the years and stress the fact that a status quo can only be properly judged when the environmental history of at least 5 preceding years is known. The fauna at the subhalocline muddy stations seems to have *recovered* well after a series of serious environmental disturbances. Fehmarnbelt station shows nevertheless a slight decline in quality that could be interpreted in connection with events in the Kattegat and the Arkona Basin. The shallower stations in Kiel Bay (above the seasonal halocline) show no major trend in the last years although there is a distinct seasonal fluctuation in most parameters.

Liibeck Bay and Bay of Mecklenburg

F. Gosselck[3]

The assessment is based on information from a national monitoring program (Fig. la). Station 7 LB (23 m) is representative for the central part of the Liibeck Bay (below 20 m). The sediment is very soft and contains hydrogen sulphide. Oxygen deficiency in the bottom water layers is *common* in late summer and autumn. The Bay of Mecklenburg is represented by station 18 MB (19 m). The sediment is sandy mud. Oxygen deficiency has been observed during the last years. The inner part of the Wismar Bay is an example of a highly polluted estuary in the Belt Sea. It is a shallow area with a maximum water depth of 6 m with a pronounced phytal character down to 4 m depth. The sediments are very fine mud. The Kadet Furrow is the link between the Belt Sea and the Baltic Sea. In this area no oxygen deficiency has been observed. The sediment is sandy mud with clay.

Lübeck Bay

In 1980 the central part of the Liibeck Bay (deeper than 20 m) was found to be colonized by a benthic community comprising a very small number of species (Zmudzinski et al. 1987). The winter recolonization phases, involving around 10 species, were followed by opportunistic communities where the abundance values were to 90% dominated by *Capitella capitata* and *Polydora ciliata* (Fig. 15). Periods of oxygen deficiency and, sometimes, hydrogen sulphide formation caused an almost total disappearance of the macrofauna, *Halicryptus spinulosus* being the only surviving species. The maximum abundance value was 2800 ind./m^2 and the maximum biomass 18 g/m^2 (wet weight)(95 % *Capitella capitata)* recorded in February 1988. No trend in either abundance values nor number of species could be observed during the period 1980-1989, although the seasonal variations were considerable.

Bay of Mecklenburg

The trends at depths greater than 20 m in the Bay of Mecklenburg were the same as in Liibeck Bay, but the effects of oxygen deficiency were less pronounced and the diversity generally higher (Fig. 16). In addition to the 14 dominant polychaete species, nine bivalves species were found. Of these, however, only *Arctica islandica* was found in all samples, but, like the other bivalves its numbers showed a decreasing tendency.

The macrobenthic diversity in the shallower regions (20 m deep or less) was relatively high 1980 and 1985-1987 (14 bivalves, 27 polychaetes, 11 other). The most common species were *Mysella bidentata, Scoloplos armiger* and *Macoma baltica*. The total biomass varied between 100 and 300 g/m^2 (*A. islandica* dominated, accounting for over 80 %) and between 5 and 20 g/m^{-2} if the bivalves were excluded. In the latter case the species *Nephtys caeca* and S. *armiger* predominated.

The number of species, abundance and biomass increased noticeably from the summer of 1987 onwards. The number of species per sample rose to over 30 and reached 41 on one occasion (Fig. 15). Abundances rose to over 4000 ind./m^2 and reached a maximum of 10000 ind./m^2. The abundance increase was caused mainly by the sudden occurrence of *Mytilus edulis* (dominance maximum 63%). This coincided with an increase in the number of *Macoma baltica* and the almost complete disappearance of *Mysella bidentata.* Particularly the increase in M. *baltica* and M. *edulis* led to higher biomass values. The values varied from 240 to 850 g/m^2 and were among the highest recorded for the Baltic Sea. The 850 g/m^2 recorded for one sample consisted mainly of bivalves (824 g); *Arctica islandica 400* g/m^2, *Mytilus edulis 240* g/m^2, *Macoma baltica 130* g/m^2 and *Astarte borealis 40* g/m^2.

In September 1988, oxygen deficient water penetrated even into the shallow parts of the Bay of Mecklenburg. Zoobenthos sampling in November 1988 showed that most of the species, found before the oxygen depletion in September, had vanished. Only the bivalves *Arctica islandica* and *Astarte borealis* survived. However, the number of species was again 24, but the composition of the fauna had changed completely, the dominant species being now *Capitella capitata* and *Pygospio elegans. Halicryptus spinulosus* was found at this site for the first time. The number of species and individuals remained low until March 1989 because the recolonization by larvae swept in from the outer parts of the Liibeck and Mecklenburg Bights in winter time was sparse in the winter 1988/89. In May 1990, 30 benthic species were found, the composition of the fauna being similar to that observed before September 1988. Except *Macoma cal carea*, all species, common for this area, were present again.

Wismar Bay

Wismar Bay is a shallow region (water depth about 6 m) with a pronounced phytal character down to a depth of 4 m. The deeper regions are covered with very fine mud.

The benthic fauna consisted of 80 species, among which *Heteromastus filiformis, Scoloplos armiger, Pygospio elegans, Mytilus edulis* and *Macoma baltica* were the most common. The mean abundance was around 12600 ind./m^2 and the mean biomass (wet weight) was about 700 g/m^2.

Since Krüger and Meyer's (1937) studies, changes in species composition, abundance and biomass in this bay have been slight. Only two deposit feeding polychaetes, *Heteromastus filiformis* and *Nereis succinea,* which have dominated the community sometimes and in some places, have been added to the list of species in the meantime (Prena 1987).

Kadet Furrow

During salt water inflows, salt-rich water from the Kattegat enters the Baltic Sea through Kadet Furrow. Consequently, past studies have shown that the diversity of the fauna there is particularly high owing to the indrift of larvae (Lowe 1963; Schulz, 1969).

Francke 1973). Matthäus (1978) estimates that the oxygen concentration at a depth of 45 m in the Arkona Basin has decreased by 0.76 ml/l during the past century.

According to HELCOM data oxygen concentrations less than 2 ml/l were measured at stations BY 1 and BY 2 from August to November and some times from May to November during the period 1983-1986.

Central Arkona Basin

Data from station BY 2 (BMP K4) (Fig. la), which is the only station with complete time series from the period 1979-1988, is used to characterise the deep central areas of the Arkona Basin.

During the investigation period, 1979-1988, the number of species have decreased (Fig. 17). During the period 1979-1983 the mean number of species was 16, during the next five year period only 11.5.

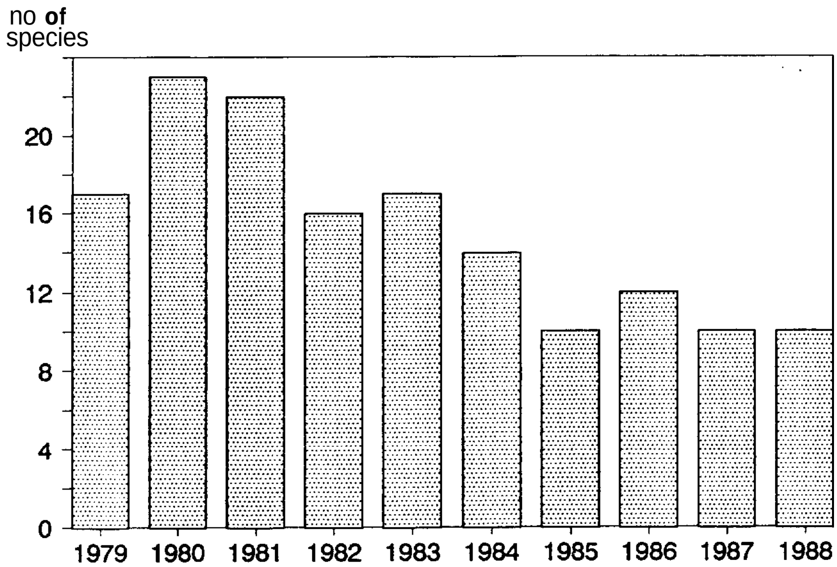


Figure 17. Changes in the number of species at station BY 2 in the Arkona Basin.

There has also been a clear change in species dominance (Table 3). Earlier common species like *Halcampa duodecimcirrata, Paraonis gracilis* and *Aricidea jeffreysii* have become very rare or even disappeared totally during 1984-1988. Also less common, but characteristic species for this area, like *Pholoe minuta, Nephtys ciliata, Polydora quadrilobata, Terebellides stroemi, Trochochaeta multisetosa* and *Pontoporeia femorata* have been absent or very rare since 1984. *Scoloplos armiger* is still numerous when it occurs, but also this species has been less common - 1979-1983 it occurred at all sampling occasions, 1984-1988 only at 50% of the sampling occasions. In general there has, since 1981, been a strong reduction in the *Macoma baltica* population, which is one of the reasons for the observed biomass decrease. On the other hand the eutrophication resistent species *Harmothoe sarsi* and *Halicryptus spinulosus* have become more important (Fig. 18).

Table 4. **Total abundance (ind/m^2) and biomass (exclusive Arctica islandica) (wet weight, g/m^2) and abundance and biomass dominance at station BY 2 in the central Arkona Basin.**

Year	date	country	Abundance Total	dominating species	Biomass Total	dominating species
1979	0610	SF	691	Scoloplos, 52%	16.71	Macoma, 42%
	1106	S	435	Scoloplos, 43%	55.41	Macoma, 93%
1980	0615	SF	8179	Capitella, 81%	30.96	Macoma, 47%
	0805	DDR	887	Diastylis, 46%	50.46	Macoma, 70%
	1113	S	1916	Scoloplos, 26%	157.11	Macoma, 70%
1981	0518	DDR	244	Scoloplos, 42%	41.35	Macoma, 77%
	0829	SF	1400	Aricidea, 27%	66.50	Macoma, 82%
	1119	S	879	Macoma, 28%	117.24	Macoma, 96%
1982	0512	DDR	122	Capitella, 17%	15.17	Macoma, 96%
	0518	S	567	hricidea, 43%	27.54	Macoma, 88%
1983	0516	DDR	54	Capitella, 50%	5.61	Macoma, 95%
	0525	SF	306	Aricidea, 21%	16.64	Macoma, 66%
	0525	S	239	Scoloplos, 33%	34.66	Macoma, 87%
1984	0507	DDR	33	Astarte e., 55%	1.34	Astarte e., 93%
	0508	S	182	Capitella, 47%	1.36	Halicryptus, 42%
	0613	SF	780	Capitella, 76%	2.72	Harmothoe, 57%
1985	0530	S	355	Scoloplos, 69%	4.42	Astarte e., 52%
	0602	SF	298	Scoloplos 72%	2.32	Scoloplos, 82%
1986	0521	S	637	Scoloplos, 75%	14.20	Scoloplos, 84%
	0526	SF	1414	Capitella, 42%	19.90	Scoloplos, 90%
	1026	DDR	77	Astarte, 42%	22.91	Astarte, 99%
1987	0523	BRD	6	Harmothoe, 100%	>0.01	Harmothoe, 100%
	0602	SF	223	Harmothoe, 54%	1.90	Priapulus, 56%
	0612	S	52	Harmothoe, 71%	0.81	Harmothoe, 96%
1988	0511	SF	400	Scoloplos, 77%	3.01	Scoloplos, 64%

Southern Arkona Basin

Benthos studies were conducted at depths between 25 an 41 m north east of the island Riigen from 1984 to 1986. Comparison of the results was possible with data collected in the same area in 1980 (Gosselck 1985) and 1957 (Löwe 1963).

Despite variations in the individual numbers of species and seasonal variations, the situation during the period from 1984 to 1986 was quite stable (Fig. 19). Between 24 and 28 species were found, the annual mean biomass (wet weight) varied from 100 to 200 g/m^2 (mean=114 g/m^2) and the annual mean number of individuals from 1000 to 2000 $ind./m^2$ (mean 1584 ind/m^2), except in July 1984, when almost 3000 $ind./m^2$ were found (Fig. 19). Peaks were observed when the populations of a few taxa increased drastically (maximum values e.g. June 1986: 8920 $ind./m^2$, including 6850 specimens of *Capitella capitata;* February 1985: 11030 $ind./m^2$ including 5520 specimens of *Mytilus edulis* and 3250 *Hydrobiides).*

5.3.4 Eastern and central part of the Southern Baltic Proper

J. Warzocha[7]

The present report is based on quantitative macrofauna samples collected within 1978-1988 at two BMP stations, Bornholm Deep, BY 5 (BMP K2) and Gdansk Deep, BCSIII-15 (BMP L1), by the Institute of Environment Protection in Gdansk (Fig. la), and at 97 stations distributed over the whole area, collected by the Sea Fisheries Institute in Gdynia (Fig. 20).

The bottom of the area studied contains three depressions: the Bornholm Basin (max. depth 90 m), the Slupsk Furrow (max. depth 90 m) and the Gdansk Basin (max. depth 110 m).

Soft mud covers areas at depths below 50 m in the Bornholm Basin, and below 70-75 m in the open Gdansk Basin and 10–30 m in the inner Bay of Gdańsk. Sediments in the shallower areas are sandy. The Slupsk Furrow is an erosion area with till and gravel in the outer parts and clay covered by mud in the central part.

Salinity in the near-bottom layer change from West to East from about 17 o/oo in the Bornholm Basin, 15 o/oo in the Slupsk Furrow to 13 o/oo in the Gdansk Basin. The stable halocline is generally formed at about 60 m in the Bornholm Basin, at 60-70 m from West to East in the Slupsk Furrow and about 80 m in the Gdansk Basin. The summer thermocline is generally formed at about 30 m, at maximum 40 m.

During the period 1978-1988 increasing oxygen deficiencies under the halocline in the Bornholm and Gdansk Basins have been observed; the areas affected being devoid of macroscopic life.

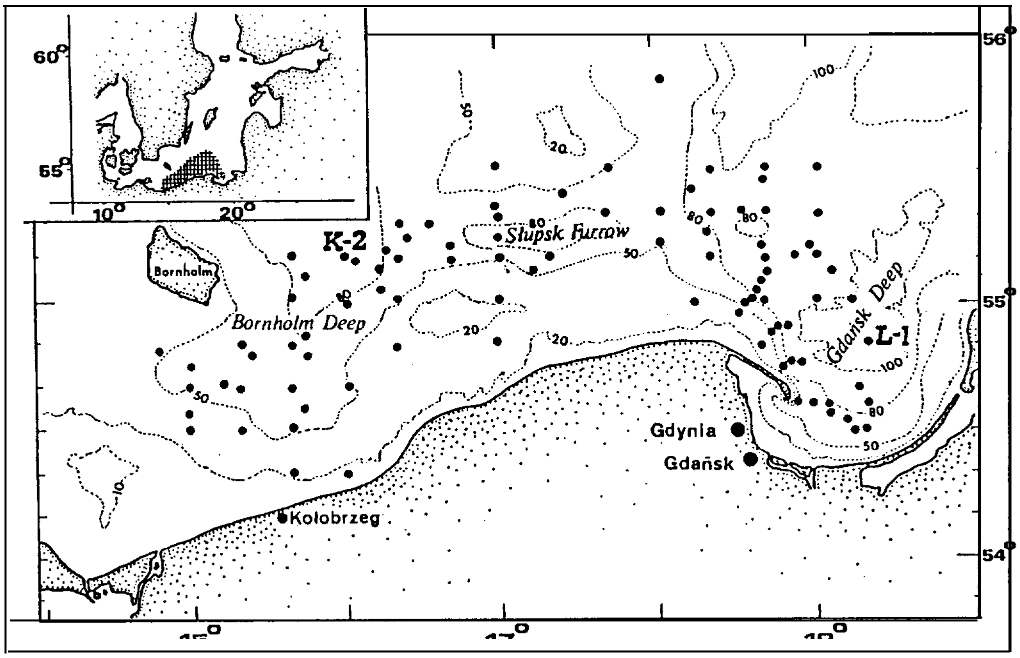


Figure 20. Area studied and stations sampled in the Southern Baltic Proper (K-2 = BY 5, L-l = BCS III 15).

In the deepest part, the Bornholm Deep (90 m), the biomass was dominated by bivalves during the first half of this century. Since the deterioration of the bottoms in the early 1960s the zoobenthos community has been dominated by polychaetes, mainly by *Scoloplos armiger* and *Harmothoe sarsi* (Zmudzinski et al. 1987). In 1979-1980 a recolonization by 4-5 species, dominated by *S.armiger* was observed (Fig. 22). From 1981 to 1983 no macrofauna was found except in September 1982 (*Diastylis rathkei, S.armiger,* Astarte *borealis, 10* ind./m^2). In 1984 a recolonization was recorded by S. *armiger (140* ind./m^2), *H.sarsi* (19 ind./m^2) and a few M. *baltica*. In 1985-1986 *H.sarsi* and S. *armiger* were found in spring, in autumn 1985 no macrofauna and in autumn 1986 only *H. sarsi.* Since June 1987 until June 1988 no macrofauna has been recorded.

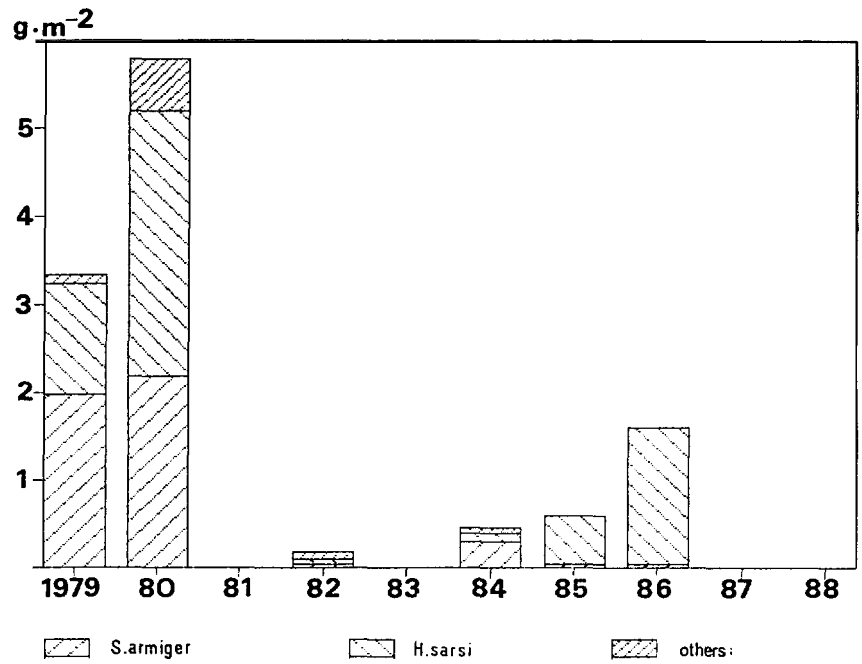


Figure 22. Changes in composition and biomass of macrofauna in the Bornholm Deep (BY 5 = BMP K 2).

Slupsk Furrow

The macrofauna inhabiting the slopes of the Slupsk Furrow does not differ from that found 40 years ago. In the deepest part only, a continuous decrease in *Astarte* spp. and increase of *Scoloplos armiger* biomass has been observed, as compared to earlier studies (Fig.22).

Gdansk Basin

In the Gdansk Basin stable macrofauna communities occurred down to 75 m. Below this depth, changes in composition and biomass of macrofauna was observed during the period 1978-1988.

Considerable changes were also observed in the northern part of the Gdansk Basin. In 1980 and 1981 macrofauna occurred down to 85 m and the bottom between 80 and 85 m was inhabited by an abundant fauna with *Scoloplos* armiger, *Macoma baltica*, and *Halicryptus spinulosus* (Fig. 24). In 1983 the complete disappearance of the bottom fauna in this area was recorded. In 1984 macrofauna, with similar composition as in 1981, but less abundant, recolonized this area. In 1985 and 1988 only S. armiger was found in abundance not exceeding 10 ind./m^2.

In the deepest part of the basin, the Gdansk Deep (110 m) the fauna was rather diverse in 1967 consisting of *Halicryptus spinulosus, Harmothoe sarsi, Scoloplos armiger* and *Pontoporeia femorata,* the biomass being 6.5 g m^2 (Fig. 25). In the period 1968-1982 no macrofauna or only semipelagic *H. sarsi* and sporadically juvenile specimens of benthic animals were recorded. In 1983 a recolonization by *H. sarsi* took place, with a maximum abundance of more than 200 ind./m^2. In 1987-1988 the bottom was again devoid of macrofauna.

In the shallow Puck Bay the biomasses values found in 1987 were considerably higher than in 1979 (396/172 g/m^2 respectively) (Osowiecki and Zmudzinski, pers.comm.)

Conclusions

Above the halocline in the Bornholm Basin, Slupsk Furrow and Gdansk Basins the fauna has been stable during the period 1979-1988. Even compared to older data changes are very small.

In the Bornholm Basin, below the halocline, the macrofauna abundance and biomass have been fluctuating very much. Periodically the deepest part has even been devoid of macrofauna.

In the deepest part of the Slupsk Furrow a shift in species composition has taken place.

In the Gdańsk Basin below the halocline an overall decrease in macrobenthos abundance and biomass has been observed 1979-1988. In the deepest part, were the fauna has been either absent or very poor since 1968, a recolonization was observed in 1983, probably as a response to an inflow of oxygenated water. In 1987-1988 the bottom was again totally devoid of macrofauna.

5.3.5 Central and Northern Baltic Proper

H. Cederwall[2]

This sub-area covers about 2/3 of the Baltic Proper. About 40 % of the area lies below the halocline (at ca 70 m) and the very deep area (below 140 m) makes up about 10 % . There is a significant difference between the eastern and western shorelines of the area. The western coast is broken with large archipelago areas and fjords, while the eastern coast is very open and contains a considerably greater proportion of shallow bottoms.

Below the halocline the sediments are, in the whole subarea, almost exclusively fine-grained, with high contents of water and organic matter. The bottoms above the halocline are significantly different in the Western and Eastern Gotland Basins. The Eastern Gotland Basin has a much higher percentage of sandy bottoms.

There are a great number of pollution sources on both sides of the subarea. Because of the natural water circulation the polluted water is concentrated to the coastal areas, and therefore the effects on zoobenthos are greater near the coasts.

Sampling stations and investigation areas are shown in Figure lb.

Eastern Gotland Basin

At station BCS III-10 (=BMP K1, 90 m), below the halocline in the southernmost part, the fauna has been rather variable since the mid-1960s (Figs. 26 & 27). Peaks in abundance and biomass were found in the periods 1965-71, 1978-81 and 1984-86. In between the fauna has been greatly impoverished. There is a negative long-term trend in both abundance and biomass. Taxa like Nemertini, *Halicryptus spinulosus, Aricidea jeffreysi, Terebellides stroemi,* and *Saduria entomon* mainly occurred in the 1960s (Fig. 26) and have (with one exception) not been found during the 1980s. On the other hand species like *Priapulus caudatus* and *Macoma baltica* were absent up to the mid 1970s. On the whole the number of taxa has decreased since the 1960s. For the period 1984-1988 the community first grew (1984-1985) but thereafter decreased (1985- 1988) (Fig. 26).

According to Warzocha (1984) the biomass in this area was lower in 1981 than in 1951-1952. Data from May 1987 (Rumohr, unpublished) shows that the fauna on 74 m, in the southernmost part, was fairly well developed compared to the situation on 90 m. In fact the results from 74 m in 1987 resemble the results found on 90 m in the late 1970s.

In the deeper parts, further north in the Eastern Gotland Basin,, no macrofauna was found in 1976 (124-240 m, Lagzdins & Persson 1981). In 1978 the fauna in the central part of the Eastern Gotland Basin was well developed down to 74 m, existed at 92 m but was not found at 130- 150 m (Persson et al. 1985). In 1980 fauna was found down to 105 m, in 1981 to 140 m but in 1982 was not found at all from 85 m down (Andersin unpubl.) and the fauna was reduced even at the 70 m-level. In 1984 though Seire (1988) found *Scoloplos armiger* at 110 m and *Harmothoe sarsi* down to 140 m in the southeastern part of the basin. One year later he found a well developed community at 83 m but only *Harmothoe* at 119-130 m and no fauna below that depth. Järvekülg and Olenin (1989) reports that on the eastern slope of the Eastern Gotland Basin the fauna was fairly well developed down to about 80 m in 1985-88, between 80 and 120 m the fauna was greatly impoverished (< 100 ind. per m^2) and below 120 m no fauna was found. This agrees well with what Persson et al. (1985) found in 1978. In 1987 fauna was found down to 150 m but in 1988 only to 130 m (Andersin unpubl.).

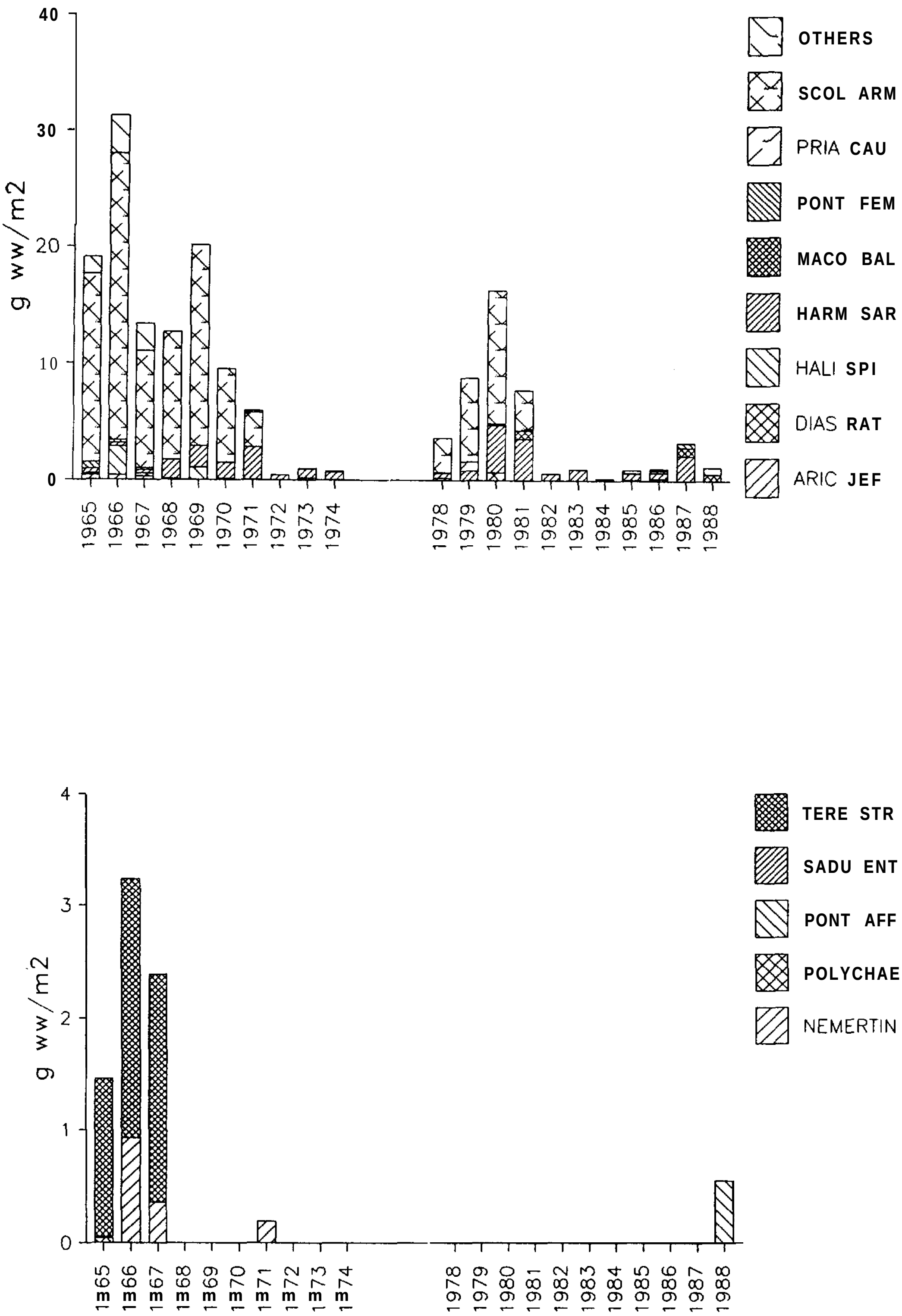
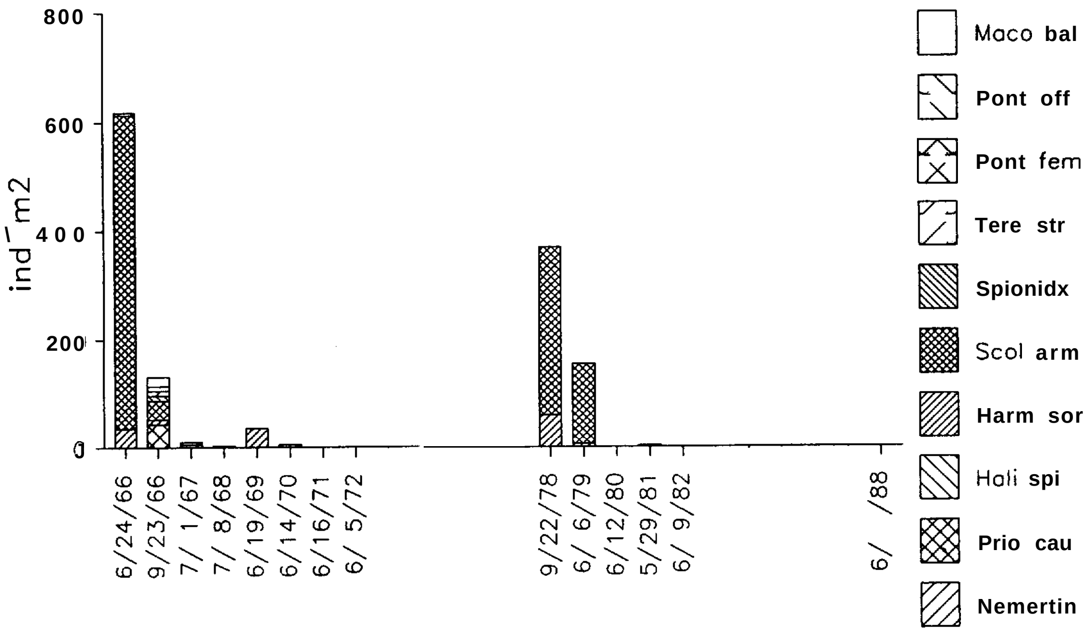


Figure 27. **Temporal variations in summed biomass (g wetweight per m^2) and biomass of different taxa at station BCS III-10 (=BMP K1), situated in the southern part of the Eastern Goland Basin at a depth of ca. 90 m.**



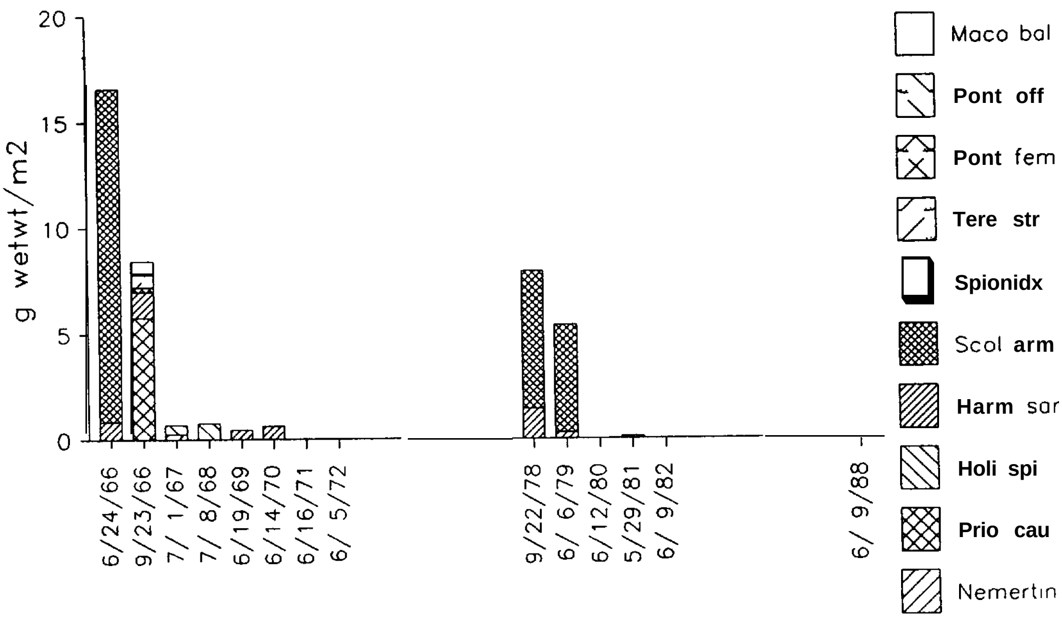
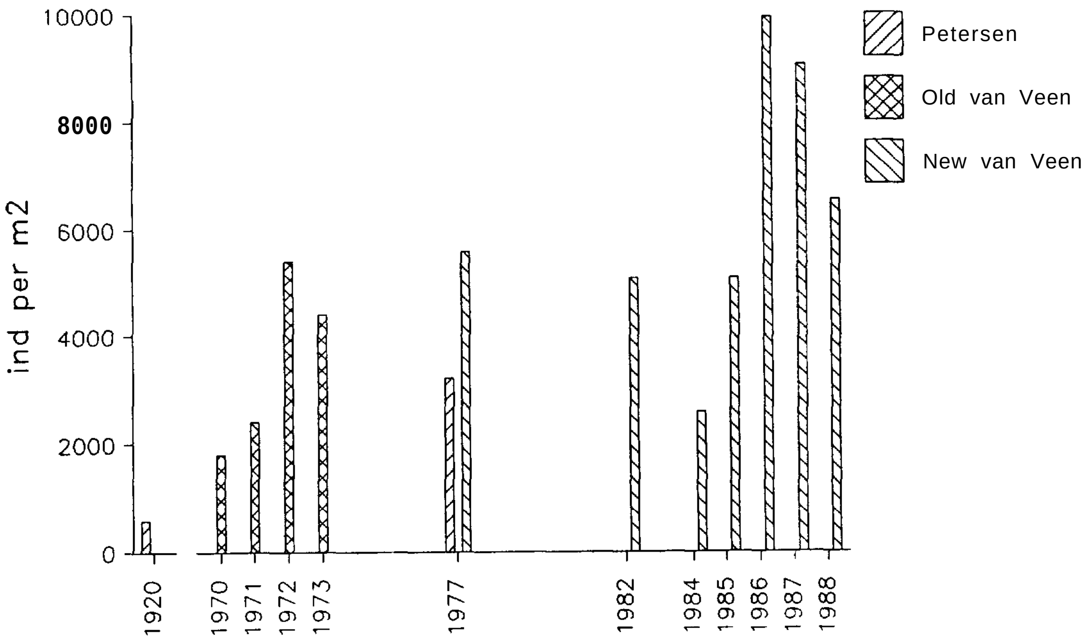


Figure 28. Temporal variations in abundance and biomass, summed and for each taxon, at station IVb 6 (110-120 m), in the central part of the Eastern Gotland Basin.



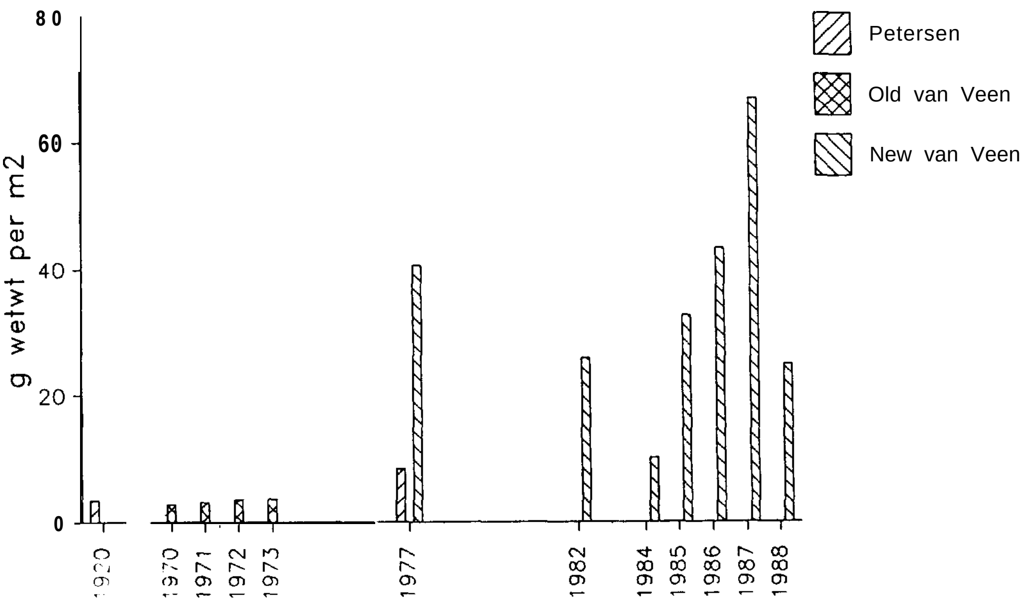


Figure 30. Temporal variations in summed abundance and biomass at station ESE När (=BMP J10), situated on the western slope of the Eastern Gotland Basin at a depth of ca. 46 m. Sampling was carried out with 0.1 m^2 Petersen grab in 1920 and 1977, with the older unmodified 0.1 m^2 van Veen-grab in 1970-1973 and with the modern 0.1 m^2 van Veen-grab (modified acc. to BMB) in 1977, 1982 and 1984-1988.

In the depth zone just below the halocline in the Landsort area (70-90 m) the number of taxa as well as abundance and biomass were very low in the 1970s and early 1980s (Cederwall 1978 and unpubl., Ankar 1986,) During the period 1984-88 the fauna and the diversity has increased (Cederwall 1988, 1989). This is an effect of higher oxygen concentrations during later years in this depth interval.

At two shallower stations (35 and 65 *m) close* to Kopparstenarna lower abundance but higher biomass was found in 1987 (Cederwall 1988) than in 1982 (Ankar 1983).

In the outer parts of the Stockholm archipelago the abundance above the halocline was cut by 50% from 1980-81 to 1985-86 (Ankar 1987). It was mainly the species *Harmothoe sarsi, Pontoporeia affinis* and *Macoma baltica* that decreased. All these changes are statistically significant, while the parallel increase in total biomass is not. It cannot be stated that the observed changes have been caused by pollution from the Stockholm area.

In the depth zone above the halocline in the Landsort area, the biomass has increased since the early 1970s, parallel to a decrease in abundance. This is caused by a shift in dominance from *Pontoporeia affinis* to *Macoma baltica* (Ankar 1986). During the 1980s there has been an increase in *Saduria entomon,* which was quite rare in the 1970s. Probably as a consequence of this increase the other predator, *Harmothoe sarsi* has decreased strongly. Although both increased biomass and the dominance switch from *Pontoporeia affinis to Macoma baltica* are generally considered to be signs of eutrophication, neither the magnitude nor other changes in species composition so far can exclude other explanations to the observed long-term changes (Cederwall 1988).

Western Gotland Basin

Also in the Western Gotland Basin there has been an increase of the fauna in the depth zone just below the halocline (75-110 m). This increase is visible in both number of taxa and abundance as well as biomass (Cederwall 1989). At station BY 38 (=BMP 11, 110 m), a station devoid of macrofauna since 1965, *Harmothoe sarsi* was found for the first time in 1985, and the population still existed in 1988 (Fig. 33). On the other hand Persson et al. (1985) found *Harmothoe* at station BY 34 (104 m) already in 1978.

Also for the Western Gotland Basin, the increase in biomass between the 1920s and the 1970s reported for bottoms above the halocline (Cederwall & Elmgren 1980), has continued into the 1980s (Cederwall 1989 and unpubl.). Similar to the trend for the Landsort area there has been a decrease in abundance in the shallower parts of the Western Gotland Basin from the mid 1970s to the early 1980s (Cederwall 1989 and unpubl.). No significant changes in species composition and dominance have taken place west of Gotland between 1976-77, 1983 and 1988 (Ankar 1984, Cederwall 1989).

Conclusions

In the deepest zone, below 140 m, there is no macrofauna, and this area has, to our knowledge, been almost completely devoid of macrofauna since the late 1960s.

For the intermediate layer, between the halocline and ca 140 m, the changes have been different in different areas. In the southern part of the Eastern Gotland Basin the fauna has been slowly impoverished since the 1960s. The salt-water inflows in 1979-80 and 1982-83 resulted in temporary increases. The reason for the disappearance, or decrease, of species like *Aricidea* and Scoloplos can possibly be decreasing salinity or magnitude of water renewal. However, there seems to be a general trend of decreasing number of species in the deeper parts of the Southern Baltic Proper (see Sections 5.33-5.34).

The situation in the central part of the Eastern Gotland Basin resembles that of the southern part, with declining benthic communities, but with generally lower figures. Also here the temporary positive effects of the mentioned salt-water inflows are evident.

In the Northern Basin and the Western Gotland Basin there has, during the 1980s, been an increase in and/or colonisation by benthos on bottoms in the intermediate depth *zone;* this is caused by the increased oxygen content of the water in the intermediate layer. That is in turn caused by an increased inflow of water from Kattegat, not saline enough to penetrate into the deepest parts of the Eastern Gotland Basin.

In the areas above the halocline increased biomass is reported from both the Eastern and Western Gotland Basins. The increase between the 1920s and the 1970s reported in the First Periodic Assessment (Zmudzinski et al. 1987) has most likely continued into the 1980s ($p<0.06$, $n=20$, Cederwall unpubl.). If this increase had been caused by longlived and slowgrowing animals being replaced by shortlived, fastgrowing ones, the biomass increase could not have been correlated with increased production. However no significant changes in dominance and species composition have occured. Therefore it seems quite reasonable to assume that macrobenthic production has increased.

5.3.6 Gulf of Riga
G. Lagzdins[5]

In the Gulf of Riga zoobenthos studies has been carried out almost continuously during the period 1945-1985, with a yearly sampling frequency of 1-3. The results show a very big biomass increase combined with a small decrease in abundance. This can be explained by an increase of molluscs and a decrease in crustaceans, e.g. *Pontoporeia* spp has disappeared in polluted coastal areas.

In the deeper central parts of the Gulf the situation was worst in the 1970s. An improvement has been observed in the 1980s, caused by greater influxes of Baltic water. At present the situation is the same as before the 1970s.

1978, 1984). The first information of deteriorated bottoms at the entrance to the Gulf of Finland originates, however, already from the 1950s (Shurin 1968) but there seems to have been a somewhat better situation again in the early 1960s, probably caused by improved oxygen conditions due to the big inflow of saline water in 1952.

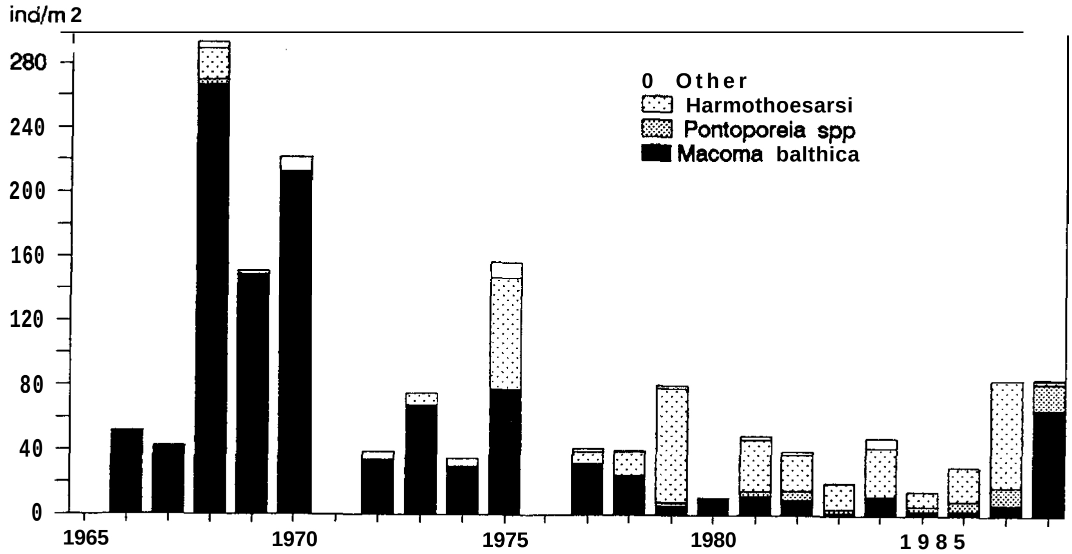


Figure 34. Variations in the total abundance and species composition at station LL 11 at the entrance to the Gulf of Finland during the period 1966-1988. (1966-1978 data according to results from the Finnish Institute of Marine Research).

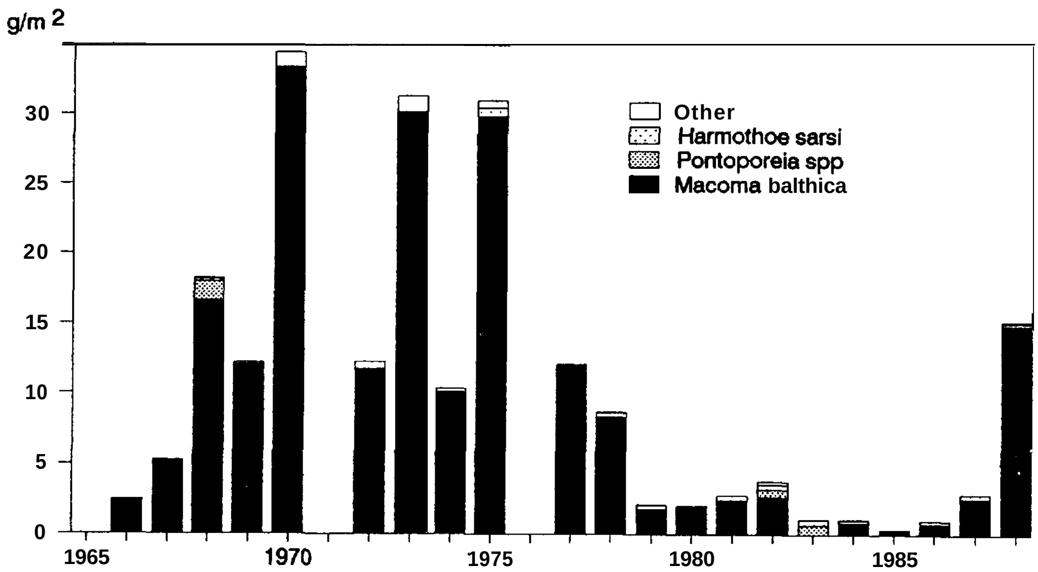


Figure 35. Variations in the total biomass (formalin wet weight) and species composition at station LL 11 at the entrance to the Gulf of Finland during the period 1966-1988. (1966-1978 data according to results from the Finnish Institute of Marine Research).

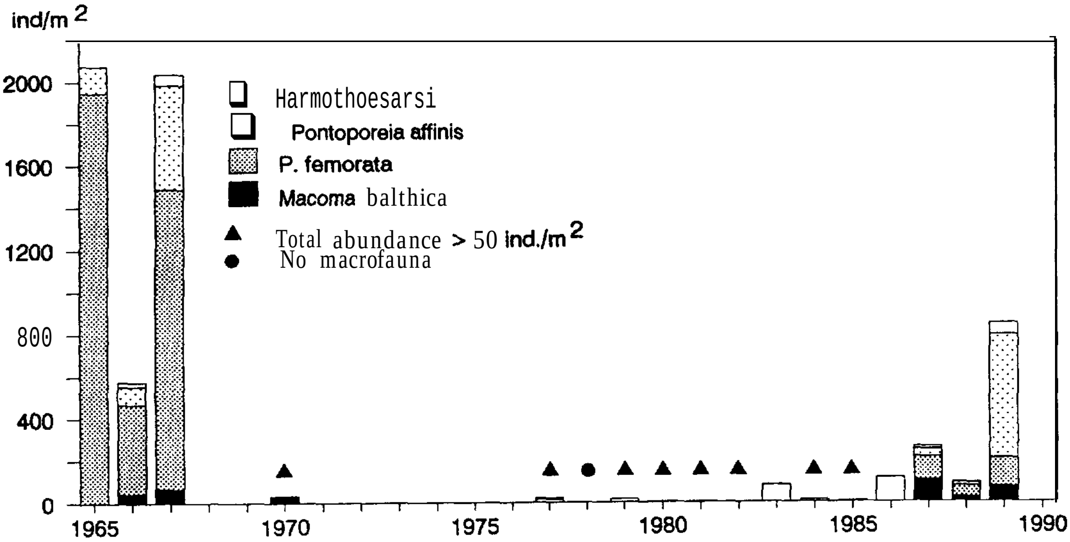


Figure 36. Variations in the total abundance and species composition at station LL **4a** in the middle part of the Gulf of Finland during the period 1965-1989. (1966-1978 data according to results from the Finnish Institute of Marine Research).

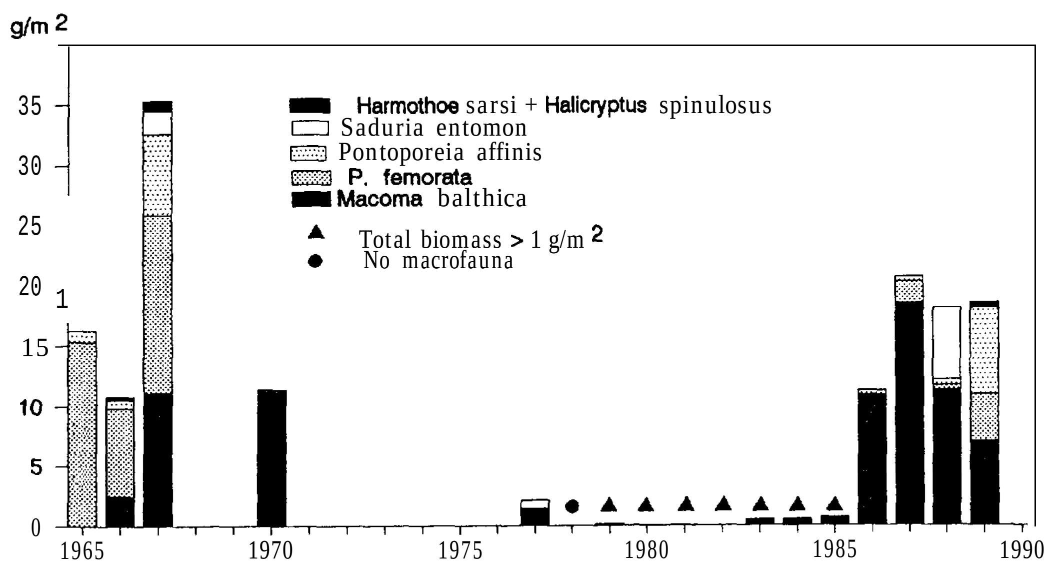


Figure 37. Variations in the total biomass (formalin wet weight) and species composition at station LL 4a in the middle part of the Gulf of Finland during the period 1965-1989. (1966-1978 data according to results from the Finnish Institute of Marine Research).

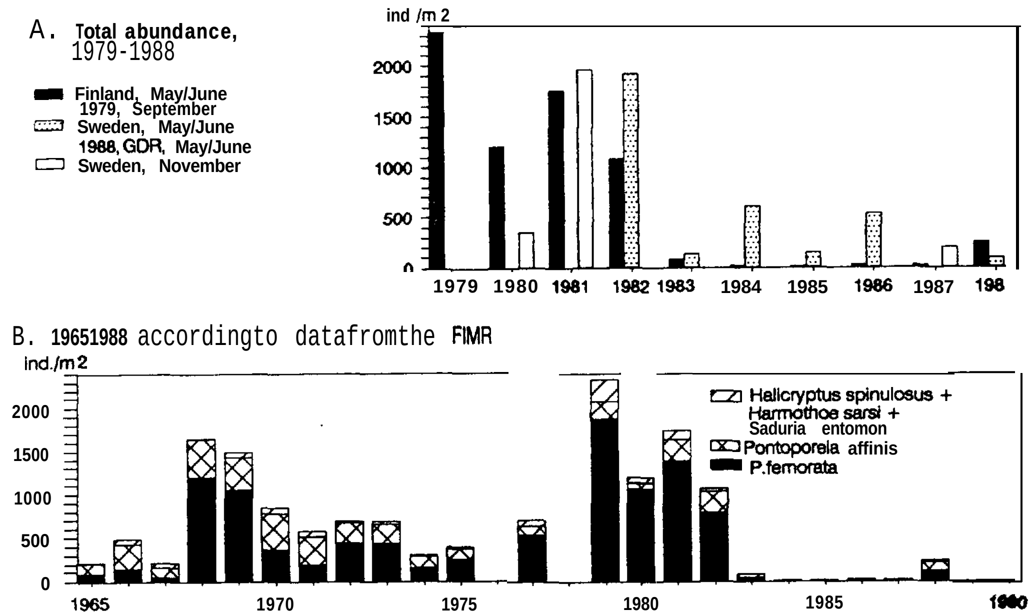


Figure 38. Variations in the total abundance during the period 1979-1988 (a) and in the total abundance including species composition during the period 1965-1988 (b) at station F 64 (280 m) in the Åland Sea.

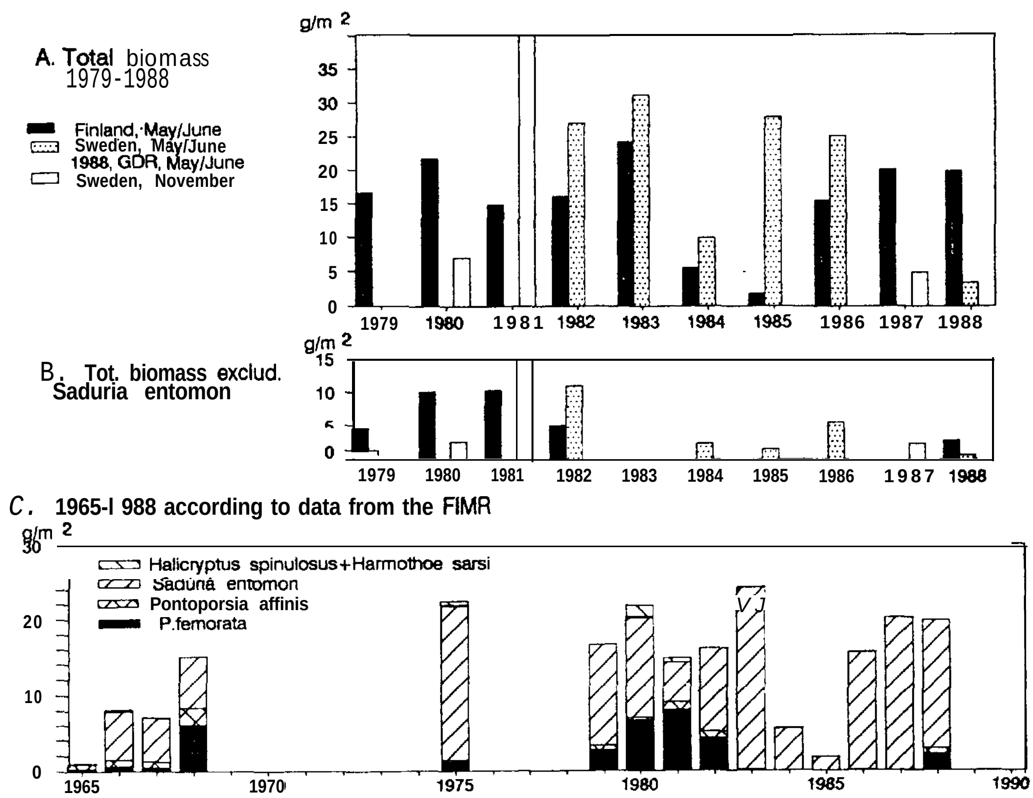


Figure 39. Variations in the total biomass (a) and the biomass exclusive *Saduria entomon* (b) during the period 1979-1988 and in the total biomass including species composition during the period 1965-1988 (c) at station F 64 (280 m) in the Åland Sea.

Both stations show the same general pattern in the fluctuations of the total abundance during the last 10 years (Figs. 40a, 42a). Minimum values are recorded in the beginning of the period and around 1984-1986, maxima 1981-1982 and 1987-1988. These fluctuations are in accordance with the long-term cyclic fluctuations described earlier from this area (Andersin et al. 1978). During the period 1979-1988 no trend can be observed in the abundance. When the results from this period is compared with older data (Figs. 40b, 42b) no trend can be detected at station US 6b. At station SR 5 some signs of an increase can be observed, both maximum and minimum values being higher during the 1980s than during 1960s and 1970s. Methodological differences in sampling and different sampling time may, however , influence the longterm results (Andersin et al. in prep.) and thus this very small difference may be negligible also in this case.

The biomass values for the period 1979-1988 show, when the big, low abundant ***Saduria entomon*** is excluded, the same pattern as the abundance values (Figs. 40b, 42b). ***Saduria entomon*** varies strongly depending on size, but at both stations ***Saduria entomon*** forms, despite size variations, about 50 % of the total biomass (Figs. 41c, 43c). Compared with data from the 1960s and 1970s no trend is detectable (Figs. 41c, 43c).

Bothnian Bay

The species composition at the two stations in the Bothnian Bay are numerically dominated by ***Pontoporeia affinis*** as in the Bothnian Sea, although the abundance values are much lower. Other species occurring are ***Prostoma obscurum,*** Oligochaeta sp. and ***Saduria entomon.*** As regards the fluctuations in abundance and biomass the two stations are not as similar as those in the Bothnian Sea.

At station C VI (=BMP A2) in the northern part of the Bothnian Bay the general pattern of the development during the period 1979-1988 resembles that in the Bothnian Sea with two abundance maxima (Fig. 44a). The maximum at the end of the period is somewhat higher than that in 1980-1981. Comparison with older data shows a quite clear increase in the total abundance during the period 1965-1988, caused mainly by an increase of ***Pontoporeia affinis*** (Fig. 44b) The same development can be seen when the biomass values are examined (Fig. 45a-c). When ***Saduria entomon*** is excluded the biomass increase is evident both within the period 1979-1988 and within the longer period 1965-1988.

At station BO 3 (=BMP A3) practically no macrofauna at all was recorded in the beginning of the period 1979-1988. In 1983 the first signs of recovery was observed, and in 1986-1988 a ***Pontoporeia*** population, very dense for Bothnian Bay circumstances, has developed (Fig. 46a). The same development is seen in the biomass values (Figs 47a-b). Only during the last years big ***Saduria entomon*** specimens have been recorded. A look at the data from 1965 onwards (Figs 46b, 47c) shows that the development up till 1977 was quite similar to that at station C VI until the break down in 1980, with typical abundance periodicity and an overall weak increase in both abundance and biomass.

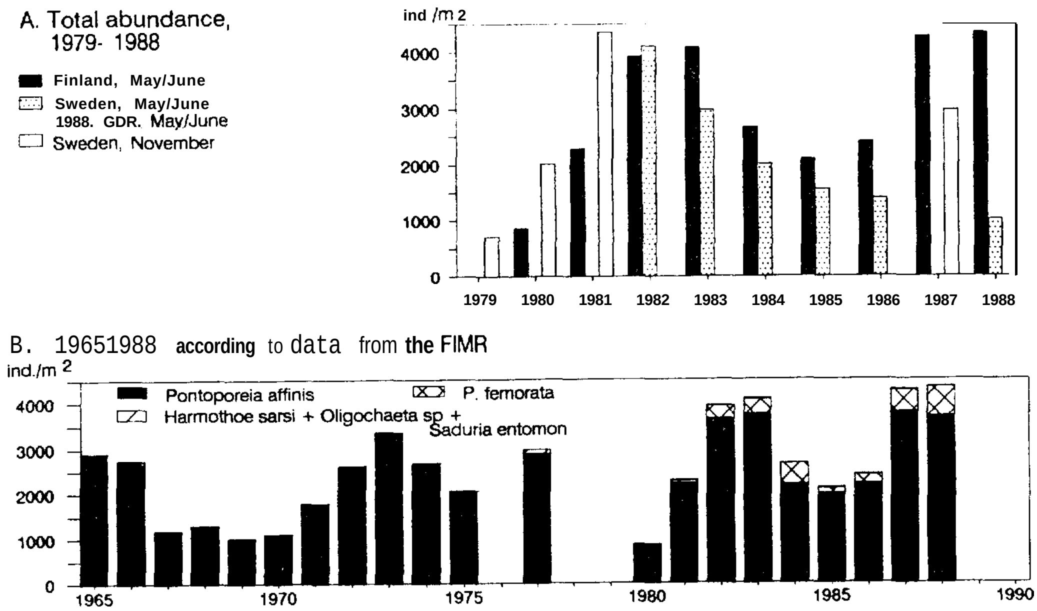


Figure 40. Variations in the total abundance during the period 1979-1988 (a) and in the total abundance including species composition during the period 1965-1988 (b) at station SR 5 (120 m) in the southern part of the Bothnian Sea.

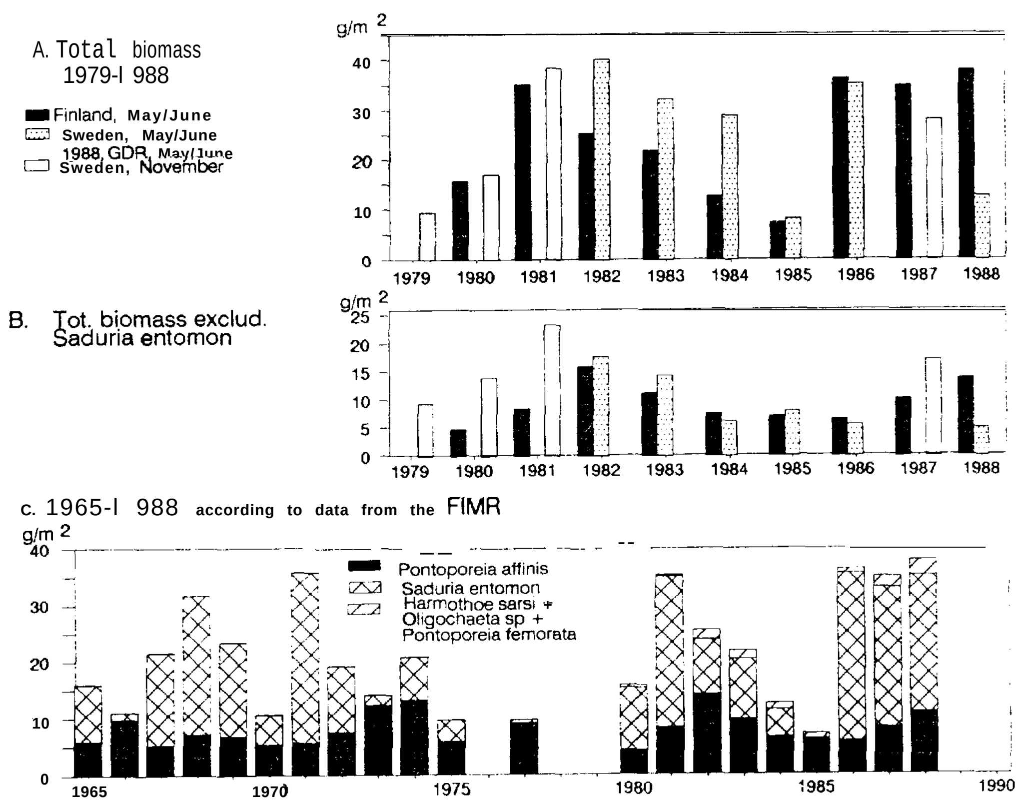


Figure 41. Variations in the total biomass (a) and the biomass exclusive *Saduria entomon* (b) during the period 1979-1988 and in the total biomass including species composition during the period 1965-1988 (c) at station SR 5 (120 m) in the southern part of the Bothnian Sea.

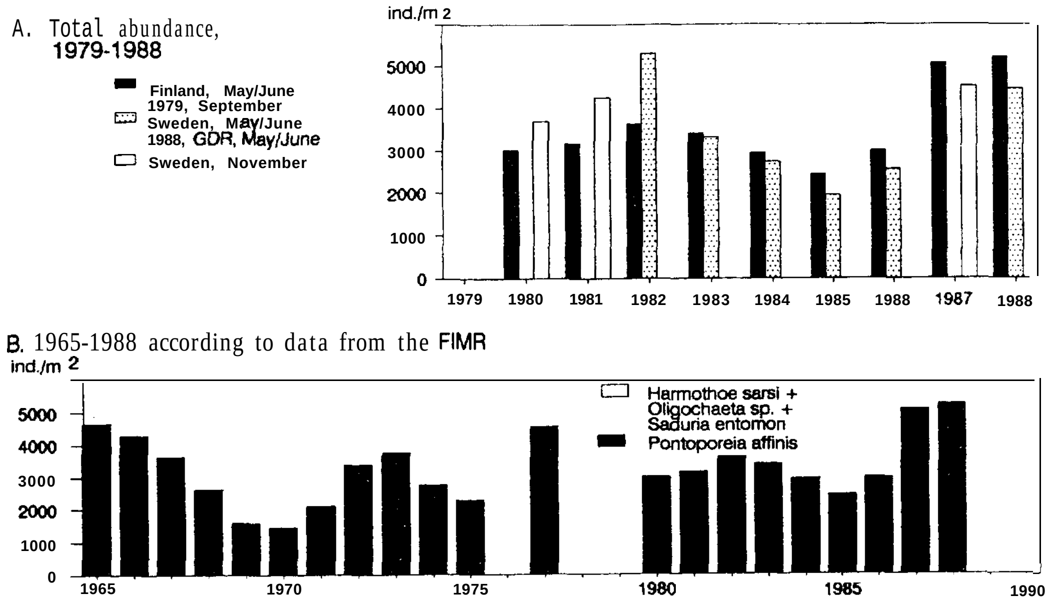


Figure 42. Variations in the total abundance during the period 1979-1988 (a) and in the total abundance including species composition during the period 1965-1988 (b) at station US 6b (80 m) in the northern part of the Bothnian Sea.

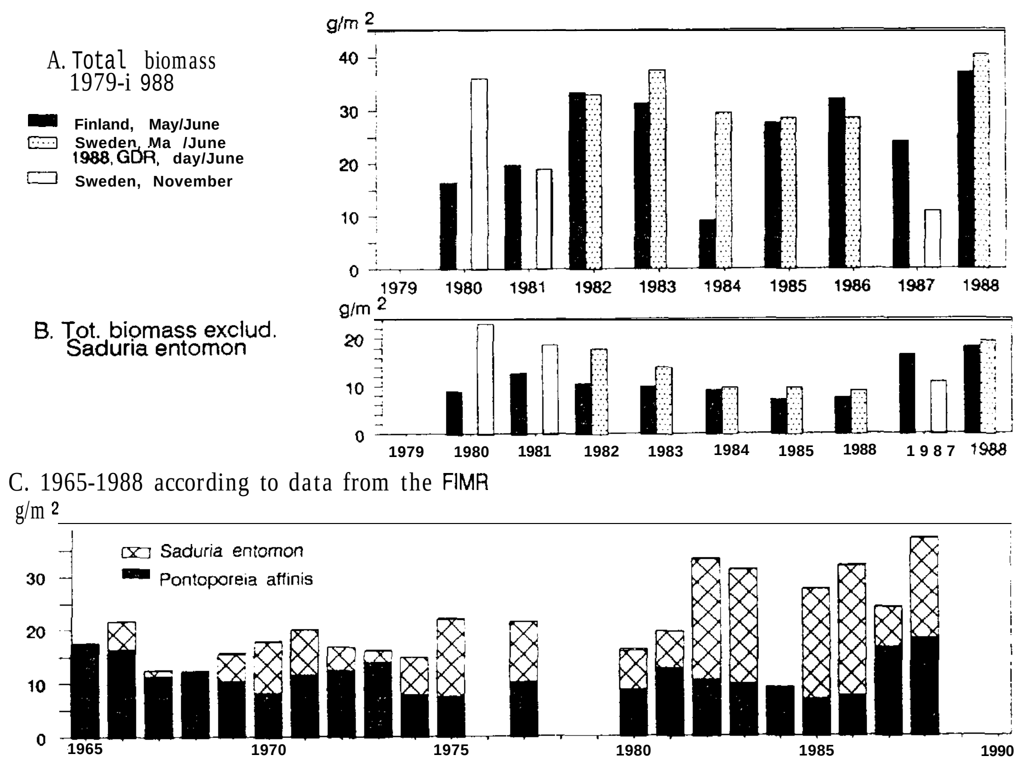


Figure 43. Variations in the total biomass (a) and the biomass exclusive *Saciuria entomon* (b) during the period 1979-1988 and in the total biomass including species composition during the period 1965-1988 (c) at station US 6b (80 m) in the northern part of the Bothnian Sea.

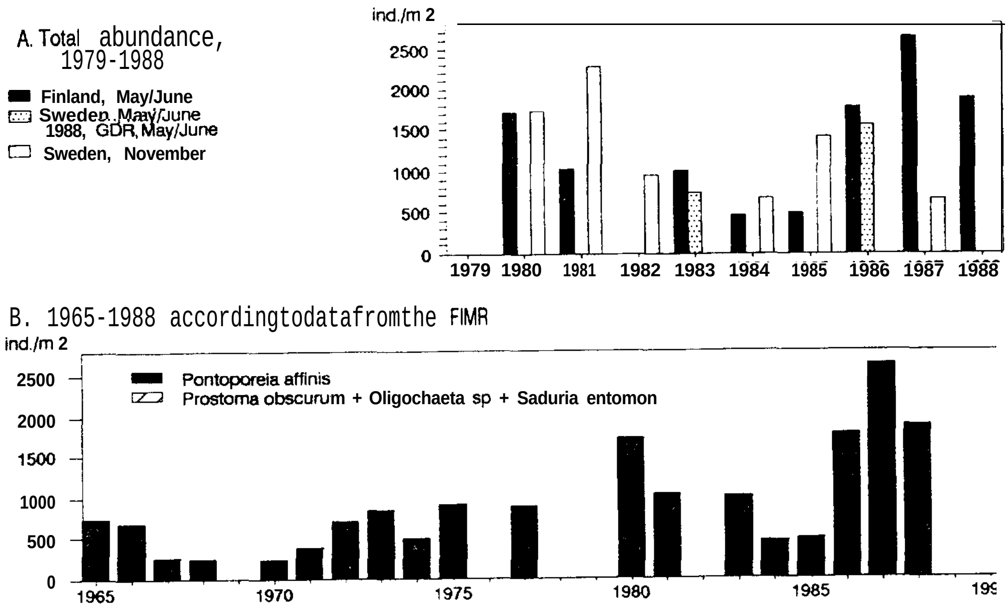


Figure 44. **Variations** in the total abundance during the period 1979-1988 (a) and in the total abundance including species composition during the period 1965-1988 (b) at station C VI (70 m) in the northern part of the Bothnian Bay.

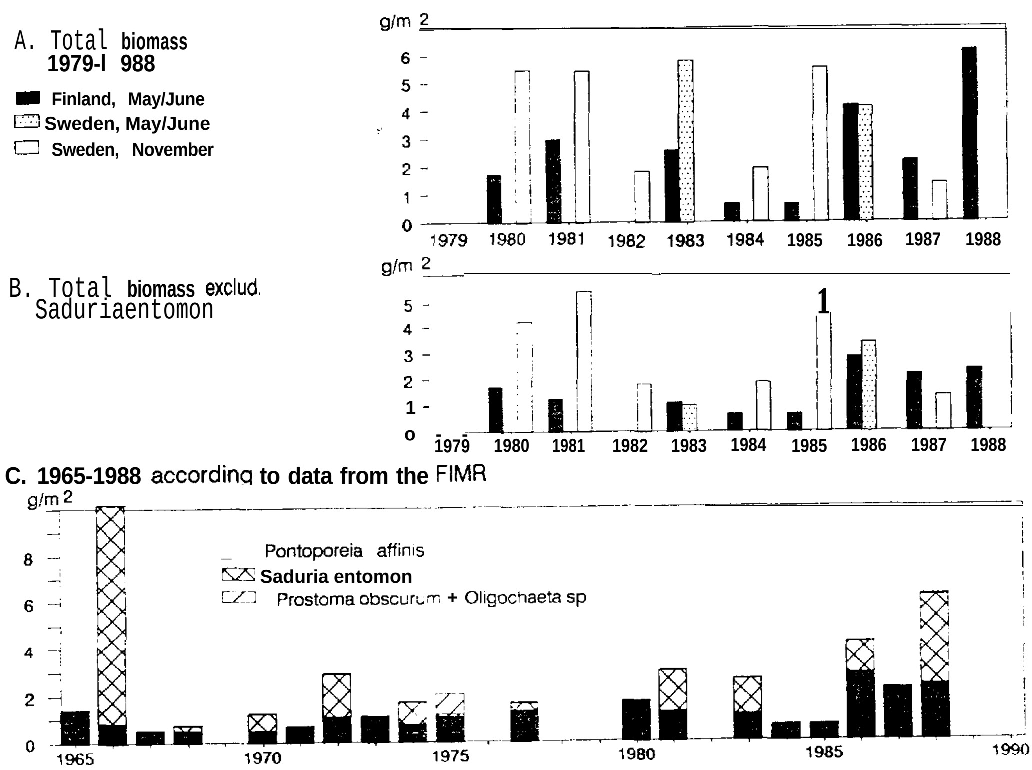


Figure 45. **Variations** in the total biomass (a) and the biomass exclusive ***Saduria entomon*** (b) during the period 1979-1988 and in the total biomass including species composition during the period 1965-1988 (c) at station C VI (70 m) in the northern part of the Bothnian Bay.

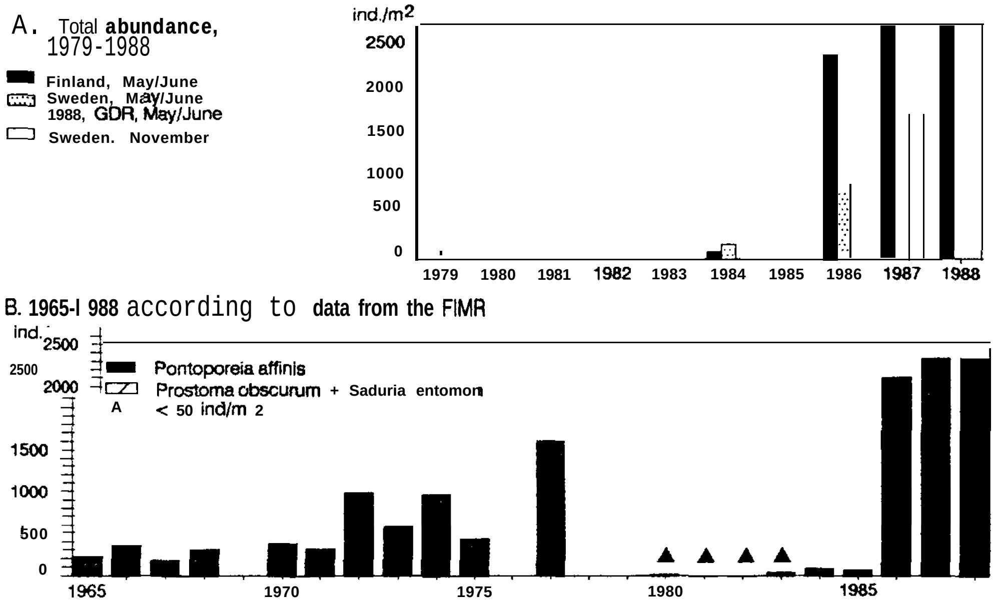


Figure 46. Variations in the total abundance during the period 1979-1988 (a) and in the total abundance including species composition during the period 1965-1988 (b) at station BO 3 (103 m) in the central part of the Bothnian Bay.

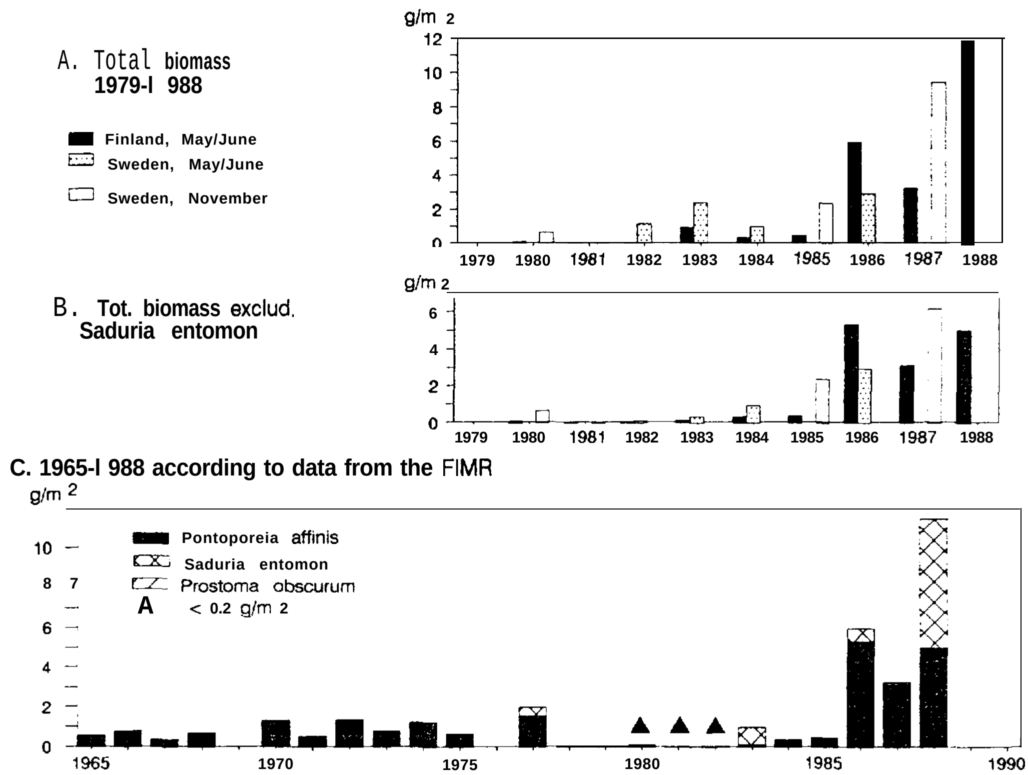


Figure 47. Variations in the total biomass (a) and the biomass exclusive *Saduria entomon* (b) during the period 1979-1988 and in the total biomass including species composition during the period 1965-1988 (c) at station BO 3 (103 m) in the central part of the Bothnian Bay.

Coastal areas

Results from areas not directly influenced by pollution are rather sparse along the coast of the Gulf of Bothnia. In the northern Bothnian Sea, in the Norrby area, benthos investigations have been carried out since the beginning of the 1970s (Leonardsson 1989). The results indicate increasing biomasses since the 1970s. Results from recent revisiting of stations sampled in the 1920s show a biomass increase of 5 times for the Bothnian Sea, with the lower confidence limit well above interannual fluctuations (Cederwall & Blomqvist, in press). Results from revisiting stations sampled 1958-1961 in the southern Bothnian Sea and Åland Sea also showed significantly higher biomass values (Cederwall & Blomqvist, in press). A comparison of the results from the 1920s, 1950s and 1980s indicates that the biomass increase mainly took place after the 1950s.

In the Bothnian Bay the nearshore stations sampled in 1986 by Cederwall and Leonardsson (1987) did not show any clear difference when compared with data from the 1970s. Neither did the results from revisiting, in 1986-87, the old stations from the 1920s reveal any significant differences (Cederwall & Blomqvist, in press).

In the Archipelago Sea fish farming has increased strongly during the last 20 years, consequently the nutrient load. Zoobenthos investigations have been carried out by different persons and institutes during the last 30 years (cf. Zmudzinski et al. 1987, Pitkänen & Kangas 1987). Jumppanen (1986) reviewed the eutrophication state of this area by means of bottom fauna and could report that the biomass had doubled during the last 20 years. Results from the Åland Archipelago show changes in the bottom fauna which can be explained by a general eutrophication of the whole area (Bonsdorff et al. 1990).

Outside pollution sources varying changes in the macrobenthic communities have been reported. A compilation of information given by Cederwall (in Bernes 1988) and by Pitkänen & Kangas (1987) show that the general trend has been an improvement during the last 20 years, visible as recolonization of earlier deteriorated bottoms and changes in species composition. The development can be seen as a response to the improvement of the techniques for treating sewage and more strict rules for sewage discharge. However, there still exists areas where no changes have taken place or the situation has grown worse.

Conclusions

In the deep areas of the Åland Sea a strong decline of the fauna has taken place during the years 1984-1988. At present there is no explanation for this decline.

In the open sea areas of the Bothnian Sea no trend in the macrobenthic abundance and biomass fluctuations could be observed during the period 1979-1988. The same was in general valid for more nearshore stations from which, however, a considerable biomass increase has been reported when values from the 1920s and 1950s has been compared with those from the 1970s and 1980s.

In the Bothnian Bay the macrobenthos at the open sea stations showed a weak continuing increase of both abundance and biomass. In the southern deep of this area (BO 3 = BMP A3) the fast recovery of a ***Pontoporeia affinis*** population could be observed, which crashed in the late 1970s.

In areas close to pollution sources the general trend in the last 15 years has been positive, although there are still areas with a negative trend. This development can be seen as a reaction on more strict rules for sewage discharge and improved techniques for sewage treatment.

SUMMARY

In the northern part of the Kattegat, the zoobenthos biomass has increased as a result of the increased deposition of biogenic organic material to the bottom.

In the southern part of the Kattegat just below the halocline, as well as in the deeper areas of the Belt Sea, a situation of reduced zoobenthos biomass and even macrofauna death, reported for the period 1979-1983, has continued during the period 1984-1988. This reduction is coupled to a more frequent occurrence of seasonal oxygen deficiency and, in some places, even the occurrence of hydrogen sulphide during the 1980s.

In the deeper parts of the Arkona Basin, a deterioration of the fauna has taken place during the 1980s. At station BY1 (K7), macrofauna death was recorded for the first time in June 1989. This decline is associated with the increased frequency of low oxygen values in the 1980s.

Below the halocline in the Bornholm Basin, the permanent bottom fauna disappeared in the 1970s. Although inflows of saline water occurred during the period 1979-1988, the periods with favourable oxygen conditions have been short and only the occasional presence of bottom fauna has been recorded. In the Gdansk Deep, a strong deterioration of the macrofauna has taken place since 1978, and the area devoid of macrofauna has increased since 1979. In the Eastern Gotland Basin, no renewal of the bottom water has occurred during the last 13 years, and the area below about 130 m has been devoid of macrofauna during the period 1979-1988.

In the Eastern Gotland Basin from the halocline (ca. 80 m) down to about 130 m there has been a clear decrease in both the abundance and biomass of macrozoobenthos during the last 25 years. In the Northern Central Basin, the entrance to the Gulf of Finland and the Western Gotland Basin, the development at this depth interval has been different. Firstly, a recovery has been recorded in areas with an impoverished fauna; secondly, bottoms formerly devoid of macrofauna have been recolonized. These changes can be explained by an improvement in the oxygen conditions.

In the middle part of the Gulf of Finland, where the fauna of the deep areas deteriorated in the 1970s, a normal macrofauna community has been re-established in the period 1986-1989.

In the areas above the halocline in the Baltic Proper, the Gulf of Riga, the Gulf of Finland and the Archipelago Sea, an increase in the zoobenthos biomass values has mainly been reported, interpreted as a response to the general eutrophication of the Baltic Sea.

In the entrance area to the Gulf of Bothnia, i.e., the Åland Sea, a decrease in both the abundance and biomass of the zoobenthos has been recorded in the deepest part of the area. In shallower areas, there is no evidence of a change.

In the open area of the Bothnian Sea, no trend has been observed during the last 20 years, but the zoobenthos values are significantly higher than those reported in the 1920s. In the open areas of the Bothnian Bay, an increase in both the abundance and biomass has been reported since 1965.

Locally polluted areas along the coast of the Gulf of Bothnia have shown a general improvement during the last 20 years, although there are some exceptions. This improvement must be seen as the result of changes in industrial processes, sewage treatment and stricter regulations. From the other regions of the Baltic Sea, the information about locally polluted areas is too scarce to draw conclusions.

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6. BALTIC FISH STOCKS

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ABSTRACT

This chapter provides a summary of the state of the commercial fish stocks in the Kattegat and Baltic Sea from 1970-1988. For stocks of cod ***(Gadus morhua),*** herring ***(Clupea harengus) ,*** and sprat ***(Sprattus sprattus),*** information is provided on yearly landings, fishing mortality (F), spawning stock biomass, total biomass, and number of recruits at age 1, as estimated from Virtual Population Analysis (VPA), for the main fish stock assessment areas in the Baltic Sea. Some information is also provided on stocks of plaice ***(Pleuronectes platessa),*** dab ***(Limanda limanda),*** sole (***Solea*** vulgaris), flounder ***(Platichthys flesus)*** and, for the Kattegat only, Norway lobster ***(Nephrops norvegicus).*** The influence of adverse environmental conditions, mainly low oxygen content in bottom waters, ***on*** the demersal fish stocks is described for each species and area affected.

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* This chapter has been prepared by the following ICES working groups: (1) Working Group on the Assessment of Demersal Stocks in the Baltic (Chairman: Dr W. Weber, Bundesforschungsanstalt fiir Fischerei, Kiel), (2) Working Group on the Assessment of Pelagic Stocks in the Baltic (Chairman: Mr B. Sjöstrand, Institute of Marine Research, Lysekil), and (3) the Division IIIa Demersal Stocks Working Group (Chairman: Ms E. Nielsen, Danish Fisheries and Marine Research Institute, Charlottenlund). Dr O. Bagge, Danish Fisheries and Marine Research Institute, Charlottenlund, also contributed to this work. Dr R. Grainger, ICES Statistician, provided initial editing of the chapter, which was compiled and finally edited by Dr J.F. Pawlak, ICES Environment Officer.

6.1 INTRODUCTION

This chapter provides a summary of the state of commercial fish stocks in the Kattegat and the Baltic Sea over the past twenty years, based on the work of three working groups under the International Council for the Exploration of the Sea: 1) the Working Group on the Assessment of Demersal Stocks in the Baltic, 2) the Working Group on the Assessment of Pelagic Stocks in the Baltic, and 3) the Division IIIa Demersal Stocks Working Group. These assessments have been made according to ICES Sub-divisions of the Baltic Sea, as shown in Figure 1.

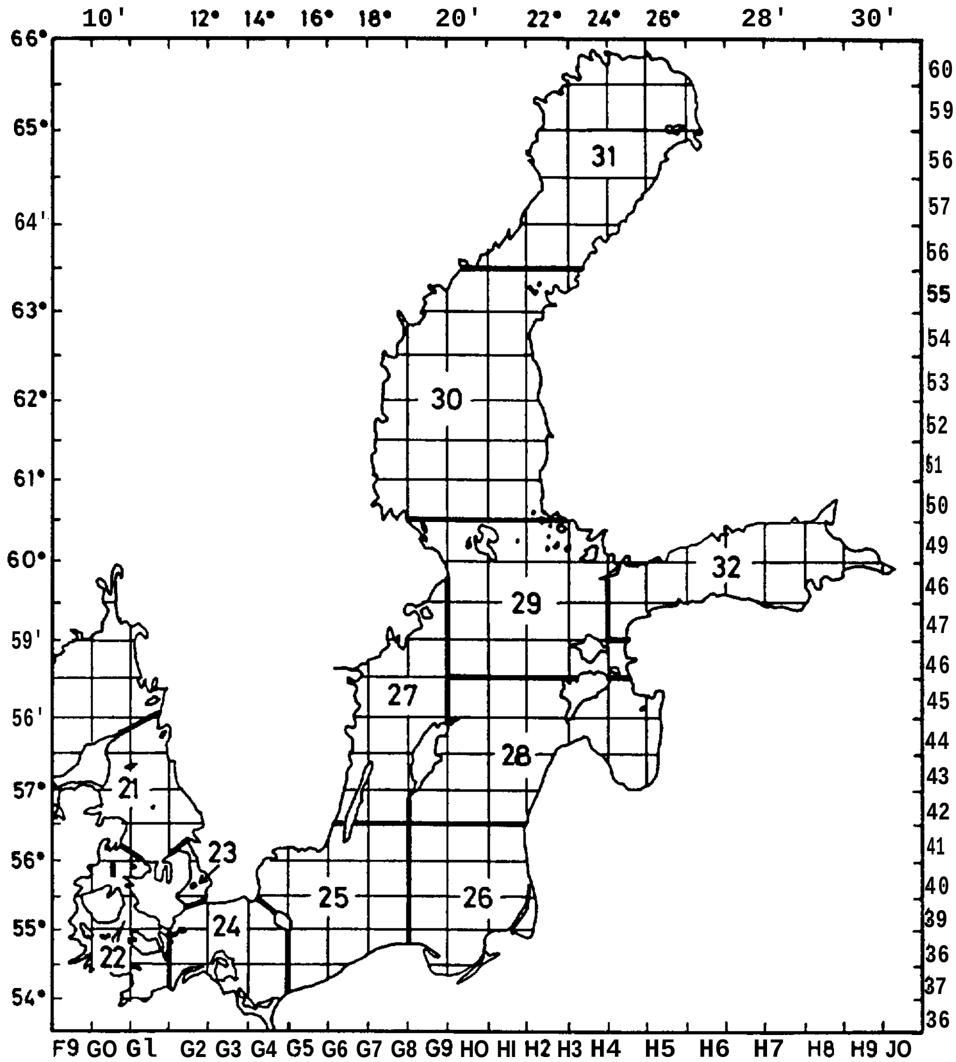

Figure 1. ICES Statistical Rectangles and Sub-divisions in the Baltic Sea. Division IIIa includes the Kattegat (Sub-division 21) plus the Skagerrak.

6.2 DEMERSAL FISH STOCKS

Information on stocks of the following demersal, i.e., bottom *or* near-bottom living, fish is provided here: cod *(Gadus morhua),* plaice *(Pleuronectes platessa),* dab *(Limanda limanda),* sole *(Solea vulgaris),* and flounder *(Platichthys flesus).* In addition, some information is also provided on Norway lobster *(Nephrops norvegicus)* in the Kattegat.

Because these species of fish live on or near the bottom of the sea, they are particularly susceptible to near-bottom low oxygen concentrations. In the central and southern Kattegat, reduced near-bottom oxygen concentrations in the autumn have been observed increasingly over the past two decades. Figure 2 shows seasonal variations in the monthly mean concentrations of oxygen near the bottom during different time periods since 1904 at or near a position 20 n.m. southeast of Anholt in the Kattegat at a depth of 30-40 m (Ertebjerg Nielsen et al., 1981), with observations for 1987, 1988, and 1989 added separately (Bagge et al., 1990). In addition to the decrease in deeper water and bottom oxygen concentrations since the period 1974-1978, the area with low oxygen concentrations during autumn has expanded also to shallow water (10-15 m) in the Kattegat (Bagge et al., 1990).

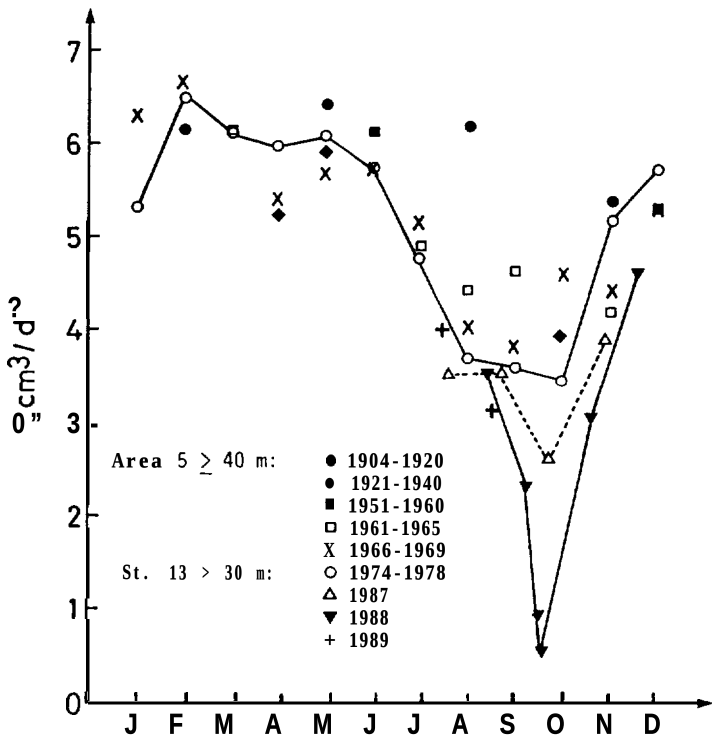

Figure 2. Seasonal variations of monthly mean oxygen concentrations near the bottom during different time periods in the eastern central Kattegat (Area 5 is Area 42G2 in Figure 1 and Station 13 is Station BMP R3).

Table 1. **Assessment of cod in the Kattegat.**

Year	Landings 1,000 t		F (3-9)		Spawn . stock biomass 1,000 t		Total biomass 1,000 t	Number of recruits in millions	
1971	15.7		0.6		29.2		66.1	37.2	
1972	17.4	17.3	0.56	0.70	37.9	35.0	68.7	22.7	25.1
1973	18.8		0.95		38.0		65.2	15.5	
1974	21.9		0.98		36.4		68.7	30.3	
1975	15.5	17.9	0.65	0.80	24.2	32.9	63.8	26.0	25.3
1976	16.3		0.78		31.6		57.9	11.0	
1977	20.1		1.04		32.4		60.9	29.5	
1978	13.4	16.8	0.83	0.83	20.7	26.5	58.4	23.4	21.2
1979	14.8		0.62		26.3		50.5	10.8	
1980	13.5		0.68		26.5		44.3	14.4	
1981	15.3	10.4	0.86	0.92	21.9	21.3	41.3	17.1	17.4
1982	12.5		1.21		15.6		38.7	20.6	
1983	12.8		1.09		15.7		40.3	20.5	
1984	11.9	12.5	1.16	1.16	16.4	16.1	36.5	11.4	13.6
1985	12.7		1.22		16.3		29.4	8.8	
1986	9.1		1.22		12.5		28.6	17.1	
1987	11.5	8.9	1.42	1.28	9.0	9.7	21.6	5.7	12.5
1988	6.1		1.20		7.7		19.2	14.7	

6.2.1 cod

Cod in the Kattegat

The development of the cod stock in the Kattegat is shown in Table 1. Yearly landings, fishing mortality (F), spawning stock biomass, total biomass, and number of recruits at age 1, as estimated from Virtual Population Analysis (VPA), are shown. Recruitment is the process by which fish enter a fishery by growing to a catchable size or migrating to an area where fishing takes place. Although recruitment to the stock has varied, applying means for three years shows a clear downward trend, with a reduction of 50% between the periods 1971-1973/1974-1976 and 1986-1988 (from 25 x 10^6 to 12.5 x 10^6) (Bagge et al., 1990).

Landinas:
Landings of cod have declined steadily since the late 1970s and during the period 1979-1988 decreased from about 15,000 t to 6,000 t (Figure 3a). The level of landings before 1979 was 13,000-22,000 t. The decrease is due to a major decline in stock size.

Recruitment and spawning stock:
The downward trend shown in recruitment during the period 1979-1988, together with an increasing fishing mortality, has resulted in the spawning stock biomass (i.e., the total weight of fish capable of reproducing) showing a decrease from 27,000 t to 8,000 t during the same period (Figure 3b).

Fishing mortality:
A distinct increase in fishing mortality has taken place, from about 0.6 in 1979 to 1.2 in 1988 (Figure 3a), due to an increased fishing effort in the Norway lobster and sole fishery in which cod has been taken as a by-catch.

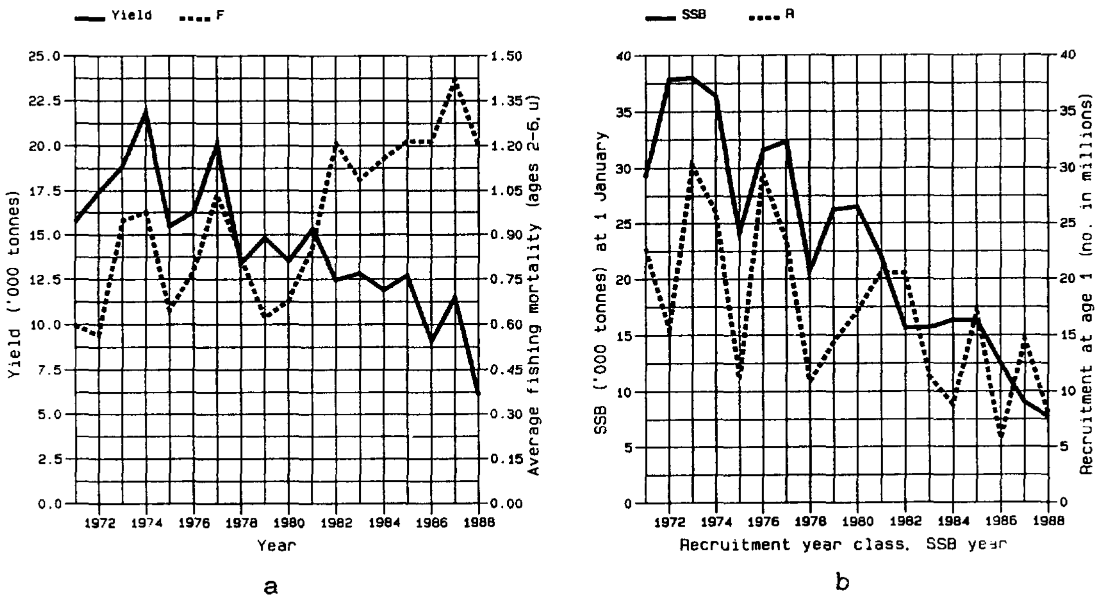

Figure 3. Cod in Sub-division 21 (Kattegat) 1971-1988.
a. Trends in yield and fishing mortality (F).
b. Trends in spawning stock biomass (SSB) and recruitment (R).

Table 2. Assessment of cod in Sub-division 22.

Year	Landings 1,000 t		F (3-7)		Spawn.stock biomass 1,000 t		Total biomass 1,000 t	No. of recruits VPA 1-group in millions	
1970	31.4		0.72		19.2		70.8	120.5	
1971	32.1	32.1	0.91	0.81	30.0	27.0	70.1	112.2	127.6
1972	32.8		0.80		31.8		82.0	150.2	
1973	38.2		0.99		34.5		83.0	33.2	
1974	31.3	33.8	1.16	1.08	37.1	32.0	70.5	151.2	85.2
1975	31.9		1.10		24.3		76.1	71.2	
1976	33.4		1.28		37.1		75.5	57.0	
1977	29.5	28.4	1.13	1.07	23.8	26.3	53.6	82.9	64.2
1978	24.2		0.80		19.0		51.1	52.6	
1979	26.0		0.76		29.9		48.1	24.8	
1980	22.9	25.1	1.06	0.96	29.9	28.8	46.5	69.4	39.4
1981	26.3		1.06		23.6		42.1	27.9	
1982	21.0		1.15		21.0		34.9	51.7	
1983	24.5	24.2	1.03	1.06	16.6	21.3	35.8	75.4	47.0
1984	27.1		1.00		26.4		36.9	13.8	
1985	22.1		1.37		26.7		32.1	9.9	
1986	12.0	15.4	1.27	1.21	14.8	19.2	19.6	47.3	21.9
1987	12.1		0.98		16.2		25.6	8.4	
1988	9.7		0.91		13.9		16.2	4.7	

Cod in the Belt Sea and Arkona Sea

The development of the cod stock in the Belt Sea (Sub-division 22) is shown in Table 2. Yearly landings, fishing mortality (F), spawning stock biomass, total biomass, and number of recruits at age 1, as estimated from VPA, are shown for each of the years from 1970-1988 and grouped as means of three years. The spawning stock biomass was relatively constant until the period 1979-1981, after which it decreased by 30%. The lowest spawning stock biomass on record was found in 1986-1988, with a reduction of 40% compared to the mean of all preceding periods. The recruitment at age 1 has dropped by about 80% since the period 1970-1972 (Bagge et al., 1990).

Landinss:
The landings in the Belt Sea and Arkona Sea (Sub-divisions 22 and 24) were constant at about 40,000-50,000 t until 1985, followed by a decline to about 18,000 t in 1989. For the period 1970-1988 they are shown in Figure 4a.

Recruitment and spawninq stock biomass:
During the period 1979-1988, recruitment in these two areas has fluctuated between 100 and 20 million at age 1 (Figure 4b). An overall declining trend has been apparent since the mid-1970s, and since the year class of 1982 all recruitment has been poor. This deterioration in recruitment is most probably due to the deterioration of environmental conditions in the bottom water layer. Berner et al. (1989) found that the lower the oxygen content is in the Belt Sea during the month of May, the lower will be the recruitment of cod during that year. The decline in recruitment, together with the maintenance of a high fishing mortality, resulted in a decrease in the spawning stock biomass from about 40,000 t in 1979 to about 22,000 t in 1988 (Figure 4b).

Fishina mortality:
Fishing mortality fluctuated around 0.9 from 1979-1985 (Figure 4a). However, it rose to a very high level in 1986, possibly due to the severe winters of 198511986 and 1986/1987, which resulted in a greater availability of cod concentrating in warmer waters in the deeper parts of the area.

Relationships with other cod stocks:
The stock of cod in the Kattegat is related to the stock in the Skagerrak and the stock in the Belt Sea and Arkona Sea (Sub-divisions 22, 23, and 24) by passive migrations of eggs and larvae and active migration of adult fish. The distribution of cod eggs and larvae and the exchange of larvae between the Kattegat and the Belt Sea seems to have changed over the past two decades (Bagge et al., 1990). It has been suggested that the decline of the cod stock in the Kattegat is caused primarily by recruitment failure in the southern Kattegat and the northern part of the Belt Sea (Sub-division 22) and by a lack of exchange of larvae with the Belt Sea and Sound (Sub-divisions 22 and 23) and, secondly, by a reduced emigration of adult cod from the Sound (Sub-division 23) and, to a lesser degree, from the Belt Sea and Arkona Sea (Sub-divisions 22 and 24) (Bagge et al., 1990).

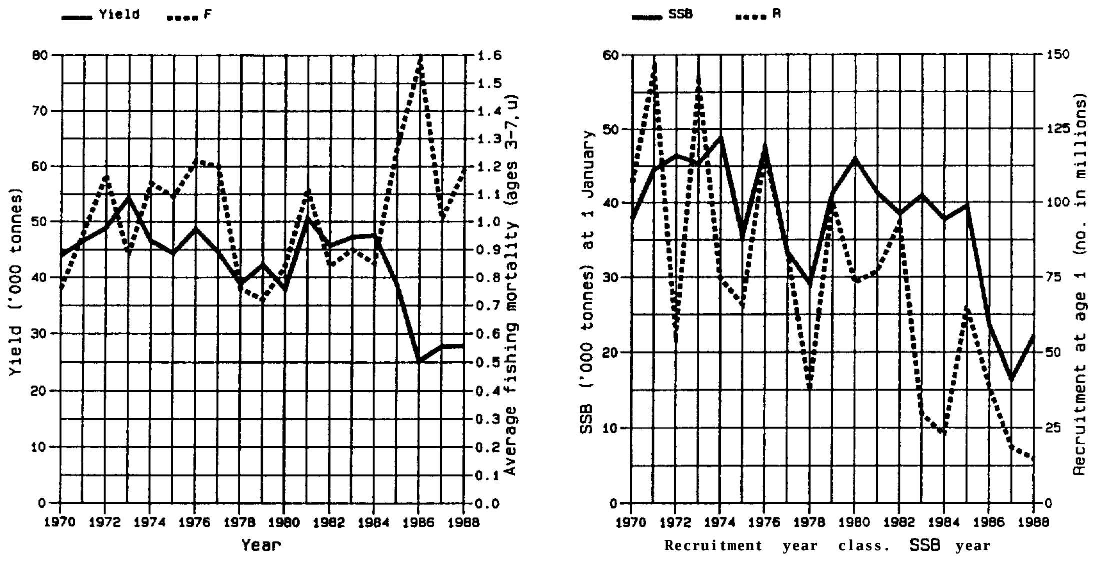

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Figure 4. Cod in Sub-divisions 22 and 24 (Belt Sea and Arkona Sea) 1970-1988.
a. Trends in yield and fishing mortality (F).
b. Trends in spawning stock biomass (SSB) and recruitment (R).

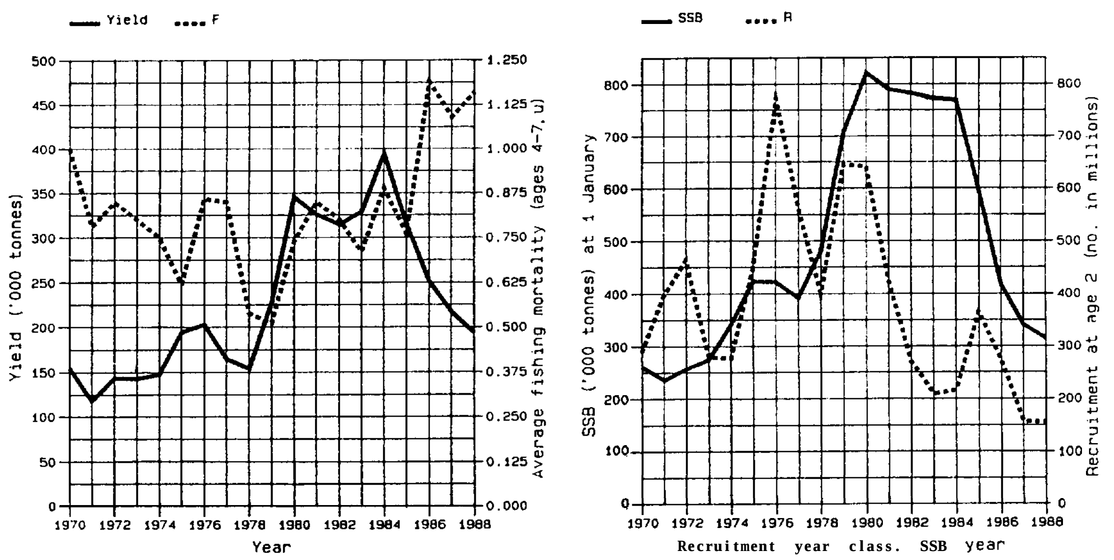

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Figure 5. Cod in Sub-divisions 25 to 32 (Baltic east of Bornholm, including the Gulfs) 1970-1988.
a. Trends in yield and fishing mortality (F).
b. Trends in spawning stock biomass (SSB) and recruitment (R).

Cod in the Bornholm Sea, Baltic Proper, Gulf of Bothnia, and Gulf of Finland (Sub-divisions 25-32)

Landinas:
From a level of about 160,000 t in the 1960s and 1970s, the landings increased to 400,000 t in 1984, the largest yield on record (Figure 5a). From 1985-1989 the landings decreased again to less than 170,000 t. The trend in landings closely reflects that in the spawning stock biomass.

Recruitment and spawnina stock:
Recruitment was very high for the year classes of 1976-1980 (Figure 5b). Year classes from 1982 onwards were of low abundance, with the exception of that of 1985.

The spawning stock increased from about 300,000 t before 1974 to about 800,000 t in 1980-1984 (Figure 5b) as a result of the strong recruiting year classes. It decreased then towards the level which had persisted prior to 1974 due to the lower recruitment and a high fishing mortality.

Fishina mortality:
Fishing mortality increased in the period 1979-1988 from 0.5 to about 1.0 (Figure 5a) due to a rise in fishing effort.

6.2.2 Plaice

Plaice in the Kattegat

The development of the plaice stock in the Kattegat, including yearly landings, fishing mortality, spawning stock biomass, total biomass, and number of recruits at age 1, as estimated from VPA, is shown in Table 3 (Bagge et al., 1990).

Landinas:
The landings of plaice decreased from 10,000 t in 1979 to less than 4,000 t in 1982, after which they have remained fairly stable at this very low level (Figure 6a). The decline in landings is due mainly to a decline in stock size, although there has also been some decline in fishing mortality (i.e., the relative numbers of fish dying as a result of being caught). The 1988 landings of about 2,000 t were the lowest recorded.

Recruitment and spawnina stock biomass:
In the 1980s, recruitment averaged only 60% of the average in the 1970s (Figure 6b). This decrease in recruitment resulted in a corresponding decrease in spawning stock biomass, which decreased from 30,000 t to 7,000 t between 1979 and 1988 (Figure 6b).

Recruitment has failed in the southern part of the Kattegat and the northern Belt Sea because of adverse environmental conditions (low oxygen content).

A decrease in the index for the yearly growth increment for plaice in the Kattegat has been observed (Bagge and Nielsen, 1987; Bagge et al., 1990).

Table 3. Plaice in the Kattegat.

Year	Landings 1,000 t		F (3-9)		Spawn.stock biomass 1,000 t		Total biomass 1,000 t	Recruits VPA l-group in millions	
1968	11.6		0.56		21.2		47.0	67.9	
1969	9.8	10.8	0.54	0.55	23.7	24.7	47.2	48.5	53.6
1970	10.9		0.56		29.2		47.7	44.3	
1971	15.0		0.88		29.7		42.5	17.1	
1972	15.8	13.7	0.73	0.72	23.6	21.8	37.3	56.8	33.2
1973	10.2		0.56		12.2		29.2	25.7	
1974	11.7		C.70		17.2		32.1	54.5	
1975	10.5	10.6	C.60	0.58	12.1	14.2	40.7	94.5	67.8
1376	9.7		C.45		13.3		46.7	54.4	
1977	11.9		0.60		30.0		47.0	28.7	
1978	13.1	11.7	C.59	0.62	27.5	25.9	36.9	17.2	la.1
1979	10.0		0.67		20.2		24.6	8.5	
1380	5.8		0.62		13.9		16.8	7.4	
1981	4.0	4.2	0.55	0.53	10.1	10.8	14.2	14.4	13.8
1982	2.9		0.42		a.4		17.0	19.5	
1983	3.5		0.52		8.1		18.3	18.5	
1984	3.6	3.5	0.74	0.53	a.3	a.5	17.2	17.3	15.5
1985	3.4		0 33		9.2		16.3	10.7	
1986	2.7		0.37		9.5		13.4	6.2	
1987	3.2	2.7	0.37	0.36	10.3	9.0	13.1	4.3	4.2
1988	2.1		0.34		7.3		a.9	2.0	

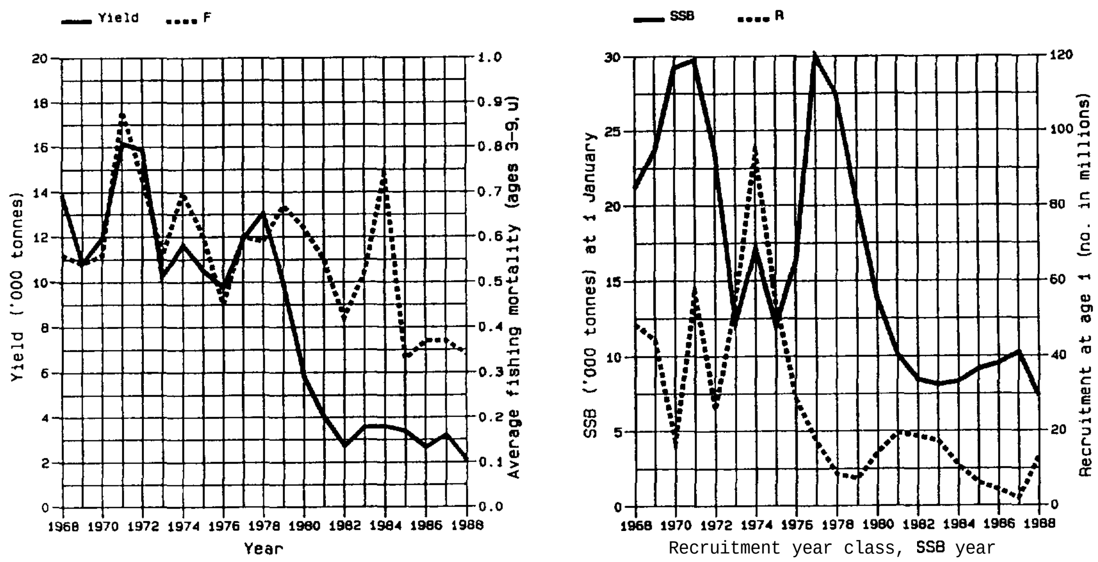

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Figure 6. **Plaice in Sub-division 21** (Kattegat) 1968-1988.
a. **Trends in yield and fishing mortality** (F).
b. **Trends in spawning stock** biomass **(SSB) and recruitment (R).**

Fishina mortalitv:
Fishing mortality was at a high level (O-45-0.75) until 1984, after which it decreased to a level of about 0.35 (Figure 6a). This was due to decreased fishing effort on this species in the southern Kattegat during summer and autumn in depths of more than 15-20 m because of low fish abundance caused by the low oxygen content.

Plaice in the Belt Sea

Plaice landings from the Belt Sea (Sub-division 22) have been decreasing since 1969, when they totalled 4,560 t; landings in 1989 were only 200 t, the lowest level since 1950. In the Arkona Sea (Sub-division 24), landings of plaice decreased from 2000 t to 300 t from 1979-1988.

Analytical assessments of the plaice stock have not been made regularly, but Figure 7 shows an assessment of spawning stock biomass and recruitment prepared by Bagge and Nielsen (1989), based on available information and certain estimates. The spawning stock biomass decreased from nearly 6,000 t in 1970 to about 1,000 t in the mid-1980s; this was caused by recruitment failure as in the Kattegat.

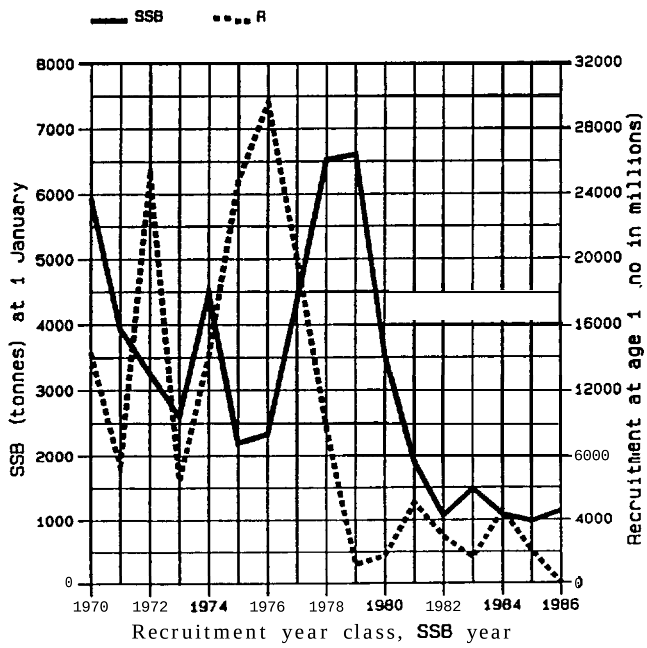

Figure 7. Plaice in Sub-division 22 (Belt Sea). Trends in spawning stock biomass (SSB) and recruitment (R), 1970-1986.

Relationships with other plaice stocks

The stock of plaice in the Kattegat is related to the plaice stock in the Skagerrak and the plaice stock in the Belt Sea by passive transport of eggs and larvae and active migration of adult fish. Larval ***surveys*** (Anon., 1989) indicate the drift of larvae from the Skagerrak to the northern part of the Kattegat and from the Belt Sea into the southern Kattegat and vice versa. Tagging experiments on adult plaice have shown that no important migration of adult plaice occurs from the Skagerrak to the Kattegat or vice versa (Bagge, 1978). However, plaice from the northern Belt Sea migrate into the southern Kattegat in the early spring (Blegvad, 1939; Simonsen et al., 1988).

6.2.3 Dab

Danish landings of dab in the Kattegat averaged about 1,700 t from 1978 to 1985, but since 1986 they have been around 1,000 t. Despite this recent decrease in landings by Danish fishermen, it appears that the population of dab in the Kattegat has increased since 1984 **(Bagge et al .,** 1990).

In the Belt Sea, Danish landings of dab have ranged from 1,500 to 1,900 t since 1979. The size of the dab stock in this area has increased since 1982 (Bagge et al., 1990).

6.2.4 Sole

Landings of sole in the Kattegat since 1975 have fluctuated between 200 and 800 t, with the lowest landings in the early 1980s. Sole landings are mainly by-catch in the Norway lobster fisheries. Sole recruitment in recent years has **shown a large increase over recruitment in the 1960s (Bagge et al., 1990).**

6.2.5 **Flounder**

No regular assessments **have been made for flounder stocks in the Baltic Sea**.

In the period 1979-1989, the international landings in the Belt Sea (Sub-division 22) decreased from about 3,000 t to about 1,300 t.

In the Arkona Sea and Bornholm Sea (Sub-divisions 24 and 25), the landings were almost constant, between 5,000 t and 7,000 t. Data from the southeastern Baltic Proper and Gulf of Gdansk (Sub-division 26) show varying catches between 2,600 t and 1,700 t.

In the northern Baltic Proper, Archipelago Sea, Bothnian Sea and Gulf of Finland (Sub-divisions 29, 30 and 32), Finnish and USSR data are available showing stable landings of about 1500 t during the period.

6.2.6 Norway lobster

The total landings of Norway lobster in the Kattegat decreased from a recent high of 2,000 t in 1984 to 1,200 t in 1988 (1989 total figures were not available). There is heavy fishing effort for Norway lobster in this area, owing to a transfer of fishing vessels from other areas and the increased use of double trawl systems. Bagge and Munch-Petersen (1979) have demonstrated that the catch of Norway lobster per hour increases with decreasing oxygen concentrations near the bottom, and that as oxygen levels decrease the percentage of females in the catch increases. Thus, the low oxygen regime makes the Norway lobster more vulnerable to the trawl fishery and reduces the protection of females carrying the eggs, which, together with the increased fishing effort, put a heavy stress on the stock of Norway lobster in this region.

The fishery has moved northward in the Kattegat in recent years and since 1988 there has been no Danish commercial fishery on Norway lobster in the Kattegat south of latitude 56°40'N.

6.2.7 Reaction to Environmental Factors and Impact on Benthos by Predation

Katteaat:
Since 1980 an increasing frequency of periods with oxygen deficit in the bottom water during spring, late summer and autumn has been observed, especially in the southern Kattegat. Values less than 2 ml/l are frequently observed in September-October.

The composition of the bottom fauna has changed towards species which are tolerant to oxygen deficit. In September and November 1988, the bottom fauna in the basin south of Anholt was depleted in depths of 30 m. In other areas, large amounts of dead echinoderms were taken by commercial trawlers.

During summer and autumn, the commercial fishing activities in the southern Kattegat are restricted to pelagic species and to demersal species in shallow water. The target species in this period was until recently the Norway lobster, but since 1986 these oxygen depletion events have caused mortality of Norway lobster and demersal fish species. Since 1988 the stock of Norway lobster has been at a very low level, making commercial fishing on that species in the southern Kattegat unprofitable. The impact on benthos due to predation is not known.

Belt Sea and Arkona Sea:
Low oxygen concentrations have been observed during summer and autumn throughout the period 1979-1988 as well as several years before. Hydrogen sulphide was observed during 1981, 1987 and 1988 in the bottom waters of the Fehmarnbelt, Liibeck Bay and Kiel Bay, and in recent years also in the southern Little Belt. Upwelling of these waters caused problems in some coastal areas.

Due to the oxygen conditiona, the commercial trawl fishery on demersal species in the deeper parts of the area is restricted to the period December-April. The spawning time for cod, plaice and flounder is between February and April at which time oxygen deficit does not occur.

The impact of cod as a predator on the benthos and macroplankton during the period 1978-1984 has been estimated. The mean biomass of invertebrates annually consumed by cod was on average 100,000 t, but it varies depending on the size of the cod stock. The ratio between vertebrates (i.e., fish) and invertebrates in the diet was found to be 2:1 (Schulz, 1987).

Bornholm Sea. Baltic Proper, Gulf of Bothnia and Gulf of Finland:
There are three main spawning areas for cod in the Bornholm Sea, Baltic Proper, Gulf of Bothnia, and Gulf of Finland (Sub-divisions 25-32), namely the Bornholm Basin, the southern Gotland Basin and the Gdansk Deep. A further spawning area of minor importance is in the Slupsk Furrow. Spawning occurs mainly from March to August, starting in the Bornholm Basin.

The cod eggs, having a specific gravity of 1,009-1,012 kg/m^3, float at certain depths depending on the salinity. In the Bornholm Deep the highest density of eggs is found at 50-70 m, whereas in the Gotland Deep and the Gdansk Deep it is at 80-90 m. The lowest salinity that keeps the eggs afloat is about 11 and the minimum oxygen content for survival of the eggs is >2.0 ml/l. In lower salinities, the eggs will sink to the bottom where normally they cannot survive. This means that the northern border for successful reproduction is found in the Gotland Deep, though the precise position of that limit may vary with salinity and oxygen.

Successful spawning seems to be positively correlated with the inflow of saline water from the North Sea, which increases the salinity in the bottom layer and improves the oxygen conditions below the halocline (Kosior and Netzel, 1989). The increasing salinity makes the cod eggs float higher in the water column where improved oxygen conditions result in improved survival of the cod eggs.

The last inflow which was followed by a major improvement in oxygen conditions was observed in 1981 and 1982. Since 1984 the oxygen conditions have improved only for short periods and only in the Bornholm Basin.

The cod stock feeds to a large degree on benthos, mainly *Saduria entomon, Harmothoe sarsi*, various amphipods and *Diastylis rathkei.* The amount of benthos eaten per year depends upon the size of the cod stock.

6.2.8 Relationship between Environmental Conditions and Life Stages of Fish

The reproduction of cod, plaice, dab, and sole shows some common features. They have pelagic eggs and pelagic larvae which take nourishment from a yolk sac the first 6-8 days after hatching before starting active feeding. Cod, plaice, and dab spawn in February-March in the Kattegat and in March-April in the Belt Sea, but dab has an extended spawning time until April-May. The early spawning times coincide with or are close to the phytoplankton spring bloom, although the dab still spawns during the period thereafter. Sole is an early summer spawner, when primary production is reduced to the summer level.

Low oxygen values in the Kattegat and Belt Sea are not found during the time of the pelagic phase of the eggs and larvae of cod, plaice, dab, and sole, and certainly not at the depths of O-15 m where the eggs and larvae are found. This means that the depletion of oxygen has no direct effect on the survival of eggs and larvae of these species in the Kattegat and Belt Sea areas (Bagge et al., 1990).

When the pelagic stage is finished after about two months, these species begin their early bottom stage in shallow water. Plaice and sole occupy very shallow sandy bottoms (0.5-2.0 m). Cod are found at 2-6 m in areas with eel grass and brown algae, and dab are found on soft muddy bottoms in areas more than 10 m deep.

Due to eutrophication, the growth of the algae ***Ectocarpus siliclosus*** and ***Chaetomorpha linum*** has increased heavily along the coasts of the southern Kattegat and in the Belt Sea between 0 and 6 m, covering the bottom like a carpet during late spring and summer (Rask, 1990). This prevents the young bottom stages of plaice and cod from feeding on Harpacticids, their main food, which may introduce extra mortality. The bottom stage of sole appears in late summer, when the algae have disappeared at depths less than 2.0 m in the Kattegat and the young bottom stages of dab settling in depths greater than 10 m are not affected.

6.2.9 Relationship **between Hypoxia and Fish Diseases**

In the Kattegat, annual investigations of diseases **in the** dab populations have been carried out each May since 1984. Three diseases were studied: (1) lymphocystis (viral origin), (2) epidermal papillomas (presumed viral origin), and (3) skin ulcerations (often associated with bacteria). During the first three years, the disease prevalence was relatively constant at a low level in the southern Kattegat. In late summer 1986, the first intensive oxygen deficiency took place in this area and has been an annual event thereafter. Since that time, the **prevalences** of the two viral diseases studied have increased (see Fig. 8) (Bagge et al., 1990).

In a non-stressed fish population, the prevalence of the viral diseases lymphocystis and epidermal **papilloma** increases with increasing age of the fish (Mellergaard and Nielsen, 1984; 1985). If a dab stock is affected by stress, e.g., by oxygen deficiency, the young fish appear to be more seriously affected by disease. Thus, a rapid spread of disease among the young age groups of dab seems to be responsible for the marked increase in disease prevalence in the southern Kattegat from 1986 to 1987 (see Figure 9) (Bagge et al., 1990). It can be expected that the disease prevalence in the dab population in the southern Kattegat will remain high until the area has been without oxygen deficiency for approximately four years, the time it takes to build a new and healthy fish stock (Mellergaard and Nielsen, 1987; Bagge et al., 1990).

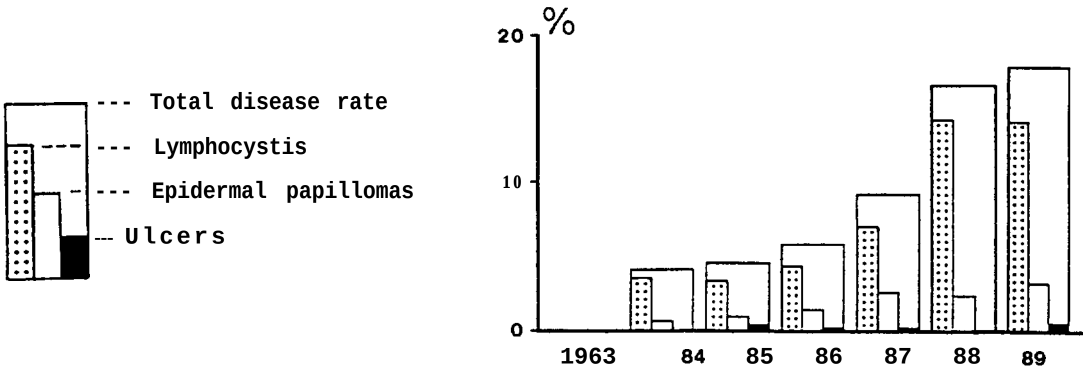

Figure 8. Development of fish diseases in dab from Sub-division 21 (Kattegat) 1984-1989 (disease prevalence in percent).

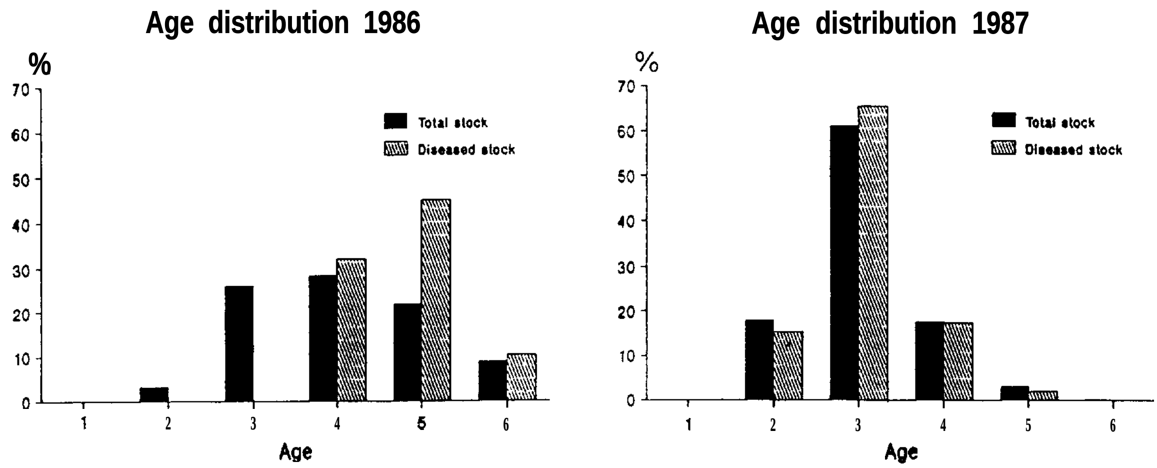

Figure 9. Age distribution of dab from Sub-division 21 (Kattegat) in 1986 (before oxygen deficiency) and in 1987 (after oxygen deficiency).

6.3 PELAGIC STOCKS

For the herring and sprat stocks in the Baltic Sea and Kattegat, the effects of environmental factors are less obvious than for some of the demersal stocks. The environment may play an important role in determining recruitment, but for all the stocks considered here, the nature of any such relationship is not yet clear. Similarly, it is not possible to draw precise conclusions as to what extent the fisheries have influenced the stock fluctuations because of limitations in the present stock data and assessment methods. Recruitment fluctuates very strongly and the influence of cod predation is not fully taken into account in the present assessment due to a lack of detailed knowledge of the interactions between the species.

6.3.1 Sprat

Sprat in the Southwestern Baltic Sea (including Sound and Belt Sea)

The area combined as an assessment unit (Sub-divisions 22 to 25) includes the shallow waters of the Belt Sea, the Sound, the Arkona Sea, and the Bornholm Sea.

The western-most parts are known to be spawning and nursery areas and may also be a place where mixing with sprat from the adjacent Kattegat takes place. The Bornholm Deep is a main spawning site of the stock.

From the early 1970s, the spawning stock of sprat diminished rather quickly from about 400,000 t to only about 19,000 t in 1980 (Figure 10b). This decrease was at least in part due to a corresponding increase in fishing mortality (Figure 10a). Within the next five years, the spawning stock biomass increased to around 150,000 t following the recruitment of the very strong 1982 and 1983 year classes (Figure 10b) and a reduction in fishing mortality. Between 1985 and 1988 the spawning stock again decreased to a level of about 40,000 t.

Catches of sprat in this assessment area rose, in response to increasing fishing mortality, to a record level in 1977 of about 36,000 t (Figure 10a) and then decreased to a level of around 14,000 t in the beginning of the 1980s. From 1985 to 1988, the yearly catch fluctuated between 15,000 t and 23,000 t. Fishing mortality was low in the beginning of the 1970s. It increased with the diminishing stock size from 0.07 in 1972 to a record high value of 1.18 in 1977 and remained high until 1980. Fishing mortality decreased to 0.23 in 1983 and since then has been fluctuating between 0.1 and 0.3.

Sprat in the Southeastern Baltic and Eastern Gotland Basin

Landings of sprat in this area (Sub-divisions 26 and 28) declined from a maximum level of about 120,000 t in 1972 to a minimum level of about 15,000 t in 1983, and subsequently they have shown an increasing trend (Figure 11a). Fishing mortality increased from 1970 to a maximum level in 1978 and then declined to a minimum in 1983. Since then, fishing mortality has remained low (Figure 11a).

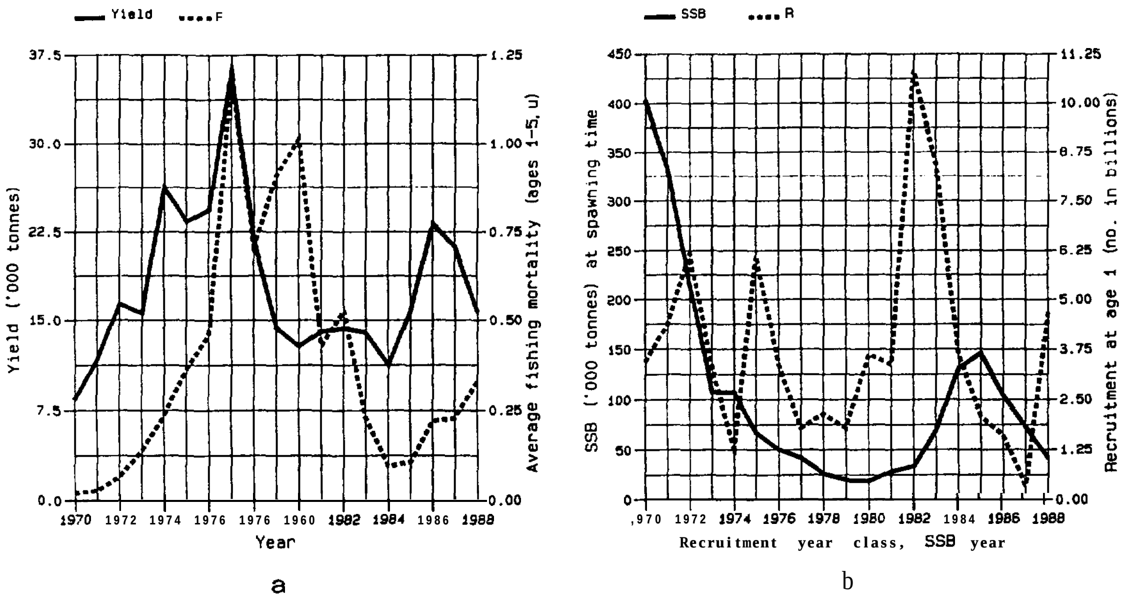

Figure 10. Sprat in Sub-divisions 22 to 25 (Belt Sea, Sound, Arkona Sea and Bornholm Sea) 1970-1988.

a. Trends in yield and fishing mortality (F).

b. Trends in spawning stock biomass (SSB) and recruitment (R).

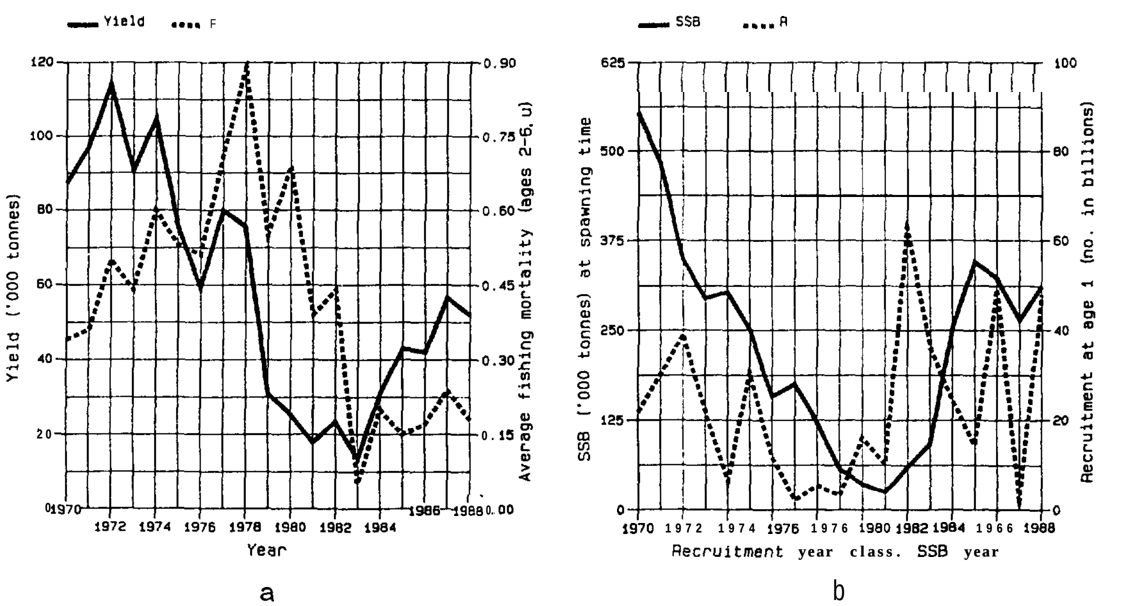

Figure 11. Sprat in Sub-divisions 26 and 28 (southern and central parts of the eastern Gotland Sea) 1970-1988.

a. Trends in yield and fishing mortality (F).

b. Trends in spawning stock biomass (SSB) and recruitment (R).

The spawning stock biomass declined rapidly from about 550,000 t in 1970 to about 25,000 t in 1981 (Figure llb). Following recruitment of the strong year class of 1982, the spawning stock increased to about 300,000 t, where it remained between 1984 and 1988.

Sprat in the Western Gotland Basin, Archipelago Sea, Gulf of Finland, and Gulf of Bothnia

The sprat stock in these areas (Sub-divisions 27 and 29-32) is exploited mainly as a by-catch in the herring fishery. Landings decreased from a maximum of about 85,000 t in 1973 to about 10,000 t in 1982, where they have remained since (Figure 12a). A decline in fishing mortality followed the decline in the landings.

The spawning stock size fell steadily from 580,000 t in 1970 to 64,000 t in 1982, and has increased only slightly since then (Figure 12b). During the 1970s there were two exceptionally strong year classes (those of 1972 and 1975), but there was no comparable recruitment in the 1980s except possibly for the 1988 year class, which may be very strong.

6.3.2 Herring

Herring in the Southwestern Baltic and the Kattegat-Skagerrak Area

This assessment unit includes the Belt Sea, the Sound, and the Arkona Sea (Sub-divisions 22-24) and spring spawners in the Kattegat and Skagerrak (Division IIIa). Herring spawn at several locations in the area, but the major spawning sites are found around the island of Riigen (Sub-division 24).

A large proportion of the herring stock migrates into the Kattegat-Skagerrak area and even into the North Sea, as indicated by tagging experiments, vertebral counts and infestation rates by ***Anisakis*** **sp.** In these waters they are exploited with autumn spawners from the North Sea, introducing large uncertainties in the actual catch of spring spawners from this assessment unit.

Catches increased from about 90,000 t in 1976 to 211,000 t in 1985, dropped to 144,000 t in 1987, but increased again to 230,000 t in 1988 (Figure 13a). Catches in the southwestern Baltic (Sub-divisions 22-24) have been rather stable in the period 1975-1988, varying between 70,000 and 110,000 t. Fishing mortality has varied between 0.6 and 1.2 in the period 1975-1988. The 1988 value was 0.76, which is approximately 40% above the level at which it is expected that the maximum sustainable yield would be taken, thus indicating that the stock is presently exploited beyond safe biological limits.

Spawning stock biomass increased from approximately 80,000 t in 1977 to 270,000 t in 1985 and has since declined to a level of about 220,000 t (Figure 13b). Recruitment has varied by a factor of 6 in the period 1977-1988. Year classes 1977, 1979, 1982, 1985 and 1986 are all comparatively abundant and that of 1986 appears to be larger than any previously observed. Recruitment in the 1980s has generally been at a level twice that observed in the 1970s.

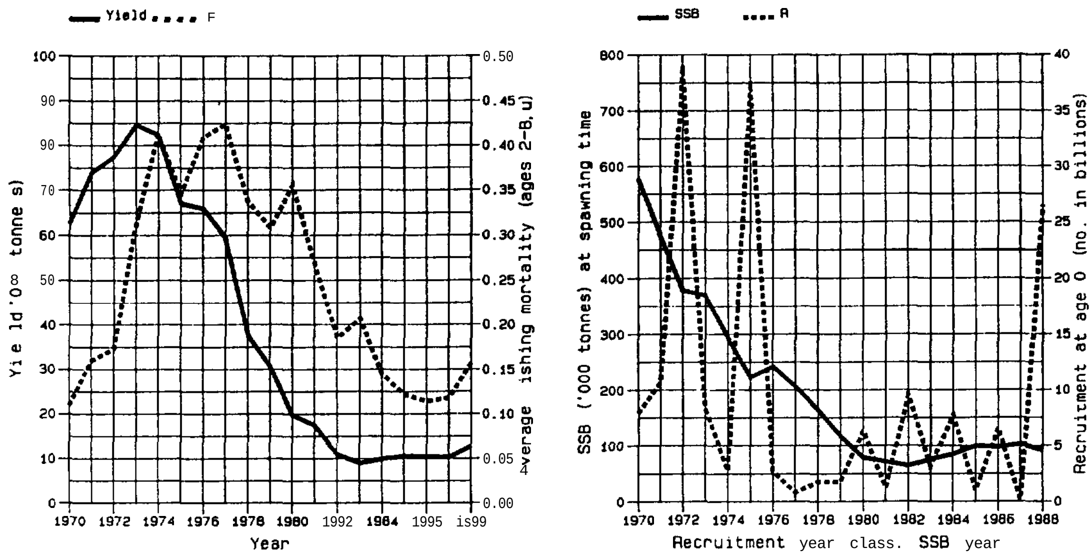

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Figure 12. **Sprat in Sub-divisions 27** (Western Gotland Sea) and 29 to 32 (Northern **Baltic Proper** and the Gulfs) 1970–1988.

a. Trends in yield and fishing mortality (F).

b. Trends in spawning stock biomass (SSB) and recruitment **(R).**

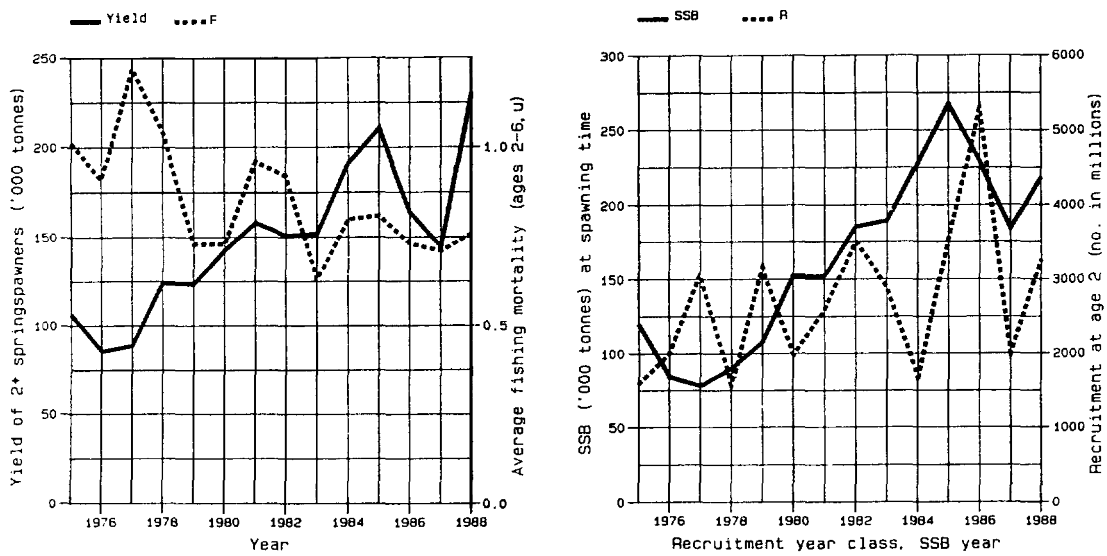

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Figure 13. **Herring** in Sub-divisions 21 to 24 (Kattegat, Belt Sea, Sound, and Arkona Sea) 1975-1988.

a. Trends in yield and fishing mortality (F).

b. Trends in spawning stock biomass (SSB) and recruitment **(R).**

Herring in the Bornholm Sea, the Southeastern Baltic and the Western Gotland Basin

Landings of herring in these areas (Sub-divisions 25 to 27) rose from 1972 to reach a maximum of nearly 200,000 t in 1979, and then declined to about 125,000 t in 1987 (Figure 14a). After an increase between 1972 and 1975, fishing mortality remained fairly steady at about 0.20 (approximately the $F_{0.1}$ level, which corresponds approximately to the most profitable level of fishing mortality) until 1984, after which it rose rapidly to about 0.38 in 1988.

The spawning stock biomass, which had been at a level of about 600,000 t for the period 1972-1980, subsequently decreased to a minimum of 260,000 t in 1988 (Figure 14b); this is partly due to a reduction in mean weights-at-age. Recruitment has fluctuated between about 3,000 and 8,000 million at age 1.

Herring in the Archipelago Sea and the Eastern Part of the Bothnian Sea

In recent years, herring catches in these areas (eastern part of Sub-divisions 29 North and 30) have been on a higher level than in the 1970s (Figure 15a). Fishing mortality has increased; the present F (0.18) is at the level of $F_{0.1}$ (0.20) which is in theory the most profitable level of exploitation in the long term. The spawning stock biomass was stable in 1974-1988 and no trend can be observed in the recruitment (Figure 15b). Thus, no significant changes took place in the state of the stock from 1974 to 1988.

Herring in the Eastern Part of the Bothnian Bay

Catches of herring in this area (eastern part of Sub-division 31) increased in the 1970s and were rather stable at about 8000 t in the 1980s (Figure 16a). Fishing mortality decreased during the 1980s; the present F (0.13) is slightly below $F_{0.1}$ (0.16). The spawning stock biomass has increased in recent years *(Figure 16b). There are major variations in the recruitment, but no trend can be observed. The effect of fishing on the stock is small.

Gulf of Riga herring

The main area of distribution of this stock is in the Gulf of Riga, with these herring making only rather limited feeding migrations into the open sea. The stock consists of two components: (1) spring spawning herring that spawn in March-July, mainly at a depth of 3-10 m on bottom vegetation, and (2) autumn spawning herring that reproduce in September to October, mainly at a depth of 10–20 m on shallows. In the first half of the 1970s, autumn herring constituted up to 45% of the total stock. In the Gulf of Riga at present, that proportion is less than 5%. The causes of the decline seem to be related to environmental conditions.

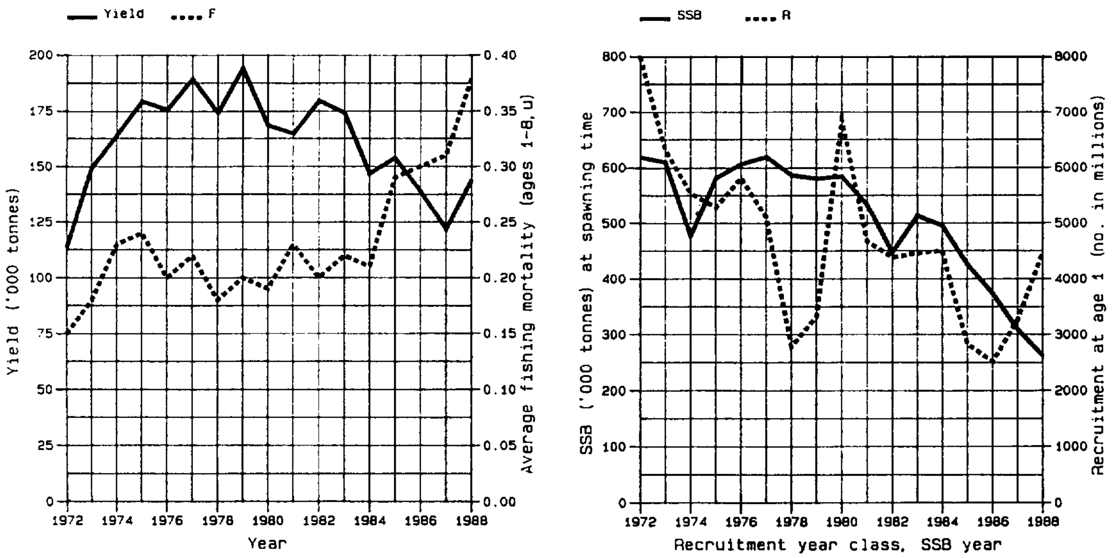

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Figure 14. **Herring in Sub-divisions 25 to 27 (Bornholm Sea, southern part of Eastern Gotland Sea, and Western Gotland Sea) 1972-1988.**

a. **Trends in yield and fishing mortality (F).**

b. **Trends in spawning stock biomass (SSB) and recruitment (R).**

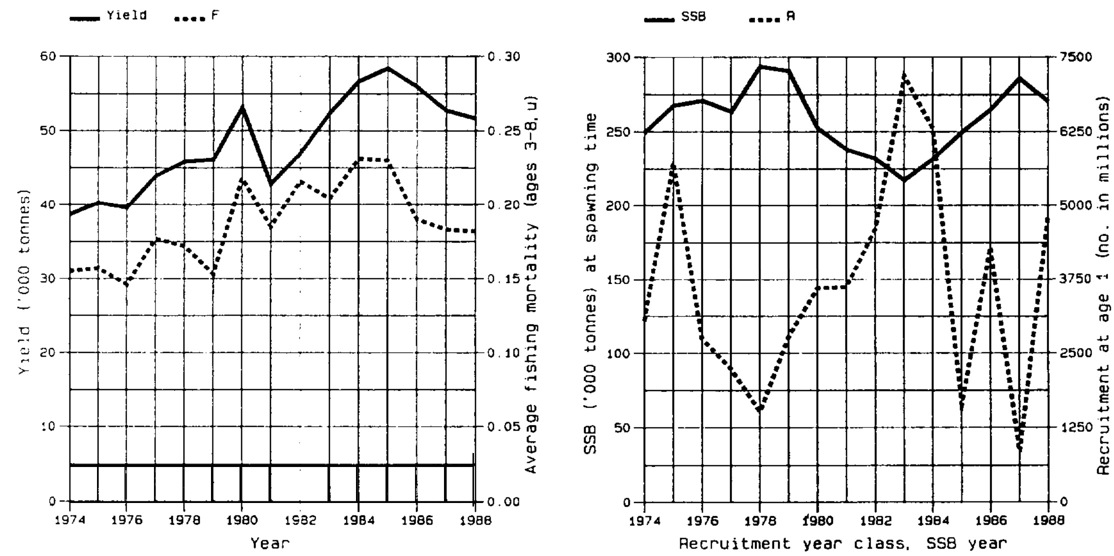

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Figure 15. **Herring in the northeastern part of Sub-division 29 (Archipelago Sea) and the eastern part of Sub-division 30 (Bothnian Sea) 1974-1988.**

a. **Trends in yield and fishing mortality (F).**

b. **Trends in spawning stock biomass (SSB) and recruitment (R).**

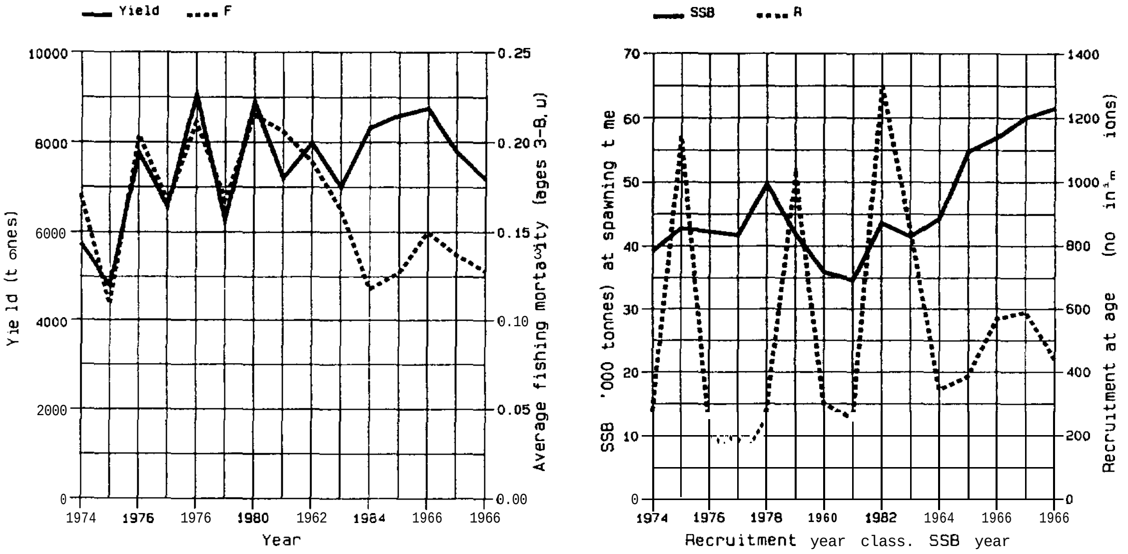

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Figure 16. Herring in the eastern part of Sub-division 31 (Bothnian Bay) 1974-1988.
a. Trends in yield and fishing mortality (F).
b. Trends in spawning stock biomass (SSB) and recruitment (R).

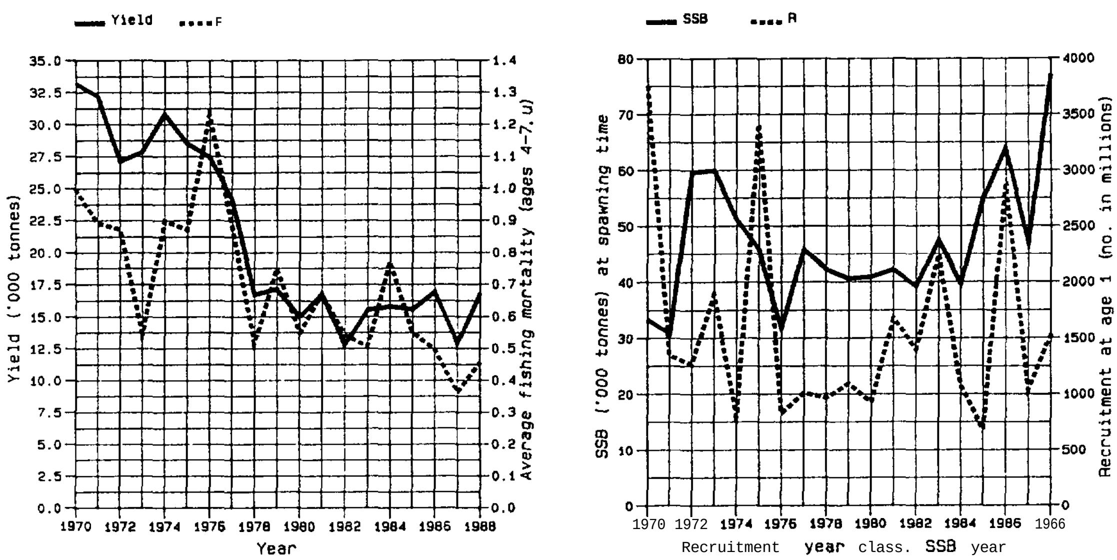

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Figure 17. Herring in the Gulf of Riga 1970-1988.
a. Trends in yield and fishing mortality (F).
b. Trends in spawning stock biomass (SSB) and recruitment (R).

Herring stock conditions depend mainly on the year class abundance, and that in turn is determined by conditions during embryonic development (mainly the structure and biomass of bottom vegetation as the spawning substrate, water temperature, etc.) and the subsequent survival rate of the larvae (probably the abundance of food, i.e., Copepod nauplii, in the period of the transfer of herring larvae to exogenous feeding). Fishing intensity has also had considerable influence on stock abundance and structure.

Owing to abundant 1970, 1975, 1983 and 1986 year classes, herring spawning stock biomass was rather high in 1972-1973 and 1986-1988, whereas at the end of the 1960s and in the second half of the 1970s, it was comparatively low (Figure 17b). The exploitation of the stock was highest in the 1970s and in the 1980s it gradually diminished (Figure 17a).

Growth changes have occurred in the Gulf of Riga herring. Weight at age was highest in 1982-1983 and lowest in 1984-1985 and 1987. Growth is mainly associated with changes in environmental conditions, e.g., severe winters and scarcity of food resources cause growth retardation.

Herring in the Eastern Gotland Basin and the Northern Baltic Proper

The stock in this area (Sub-divisions 28 and 29 South) seems to have rather migratory behaviour. It undertakes spawning migrations into the Gulf of Riga, the Gulf of Finland and probably the Archipelago Sea, as well as feeding migrations into the areas of neighbouring stocks.

In the 1970s and 1980s, the best year classes of herring in these sub-divisions hatched in 1975, 1981, 1983 and 1986, and the poorest in 1984, 1976, 1978, 1985 (Figure 18b). The spawning stock biomass was almost constant at about 200,000 t during the observation period.

The stock has been fished rather moderately. After being in excess of 0.30 in 1972 and 1973, fishing mortality has subsequently fluctuated between 0.18 and 0.26 (Figure 18a). Because the stock has remained fairly stable, the trend in catches closely reflects that in fishing mortality.

Important changes have been noted in the average weight at age of the stock. During the period 1977-1988, the maximum weights were observed in 1980-1981. A serious decline followed in 1983-1985. Only in 1988 did some increase in mean weight of younger age groups become apparent. Investigations to relate the increasing winter concentrations of nitrate and phosphate in the surface water layer of the Baltic Proper with fish stocks showed no clear relationship, although in general a 5% increase in total fish biomass between 1973 and 1984 was estimated (Nehring et al., 1989).

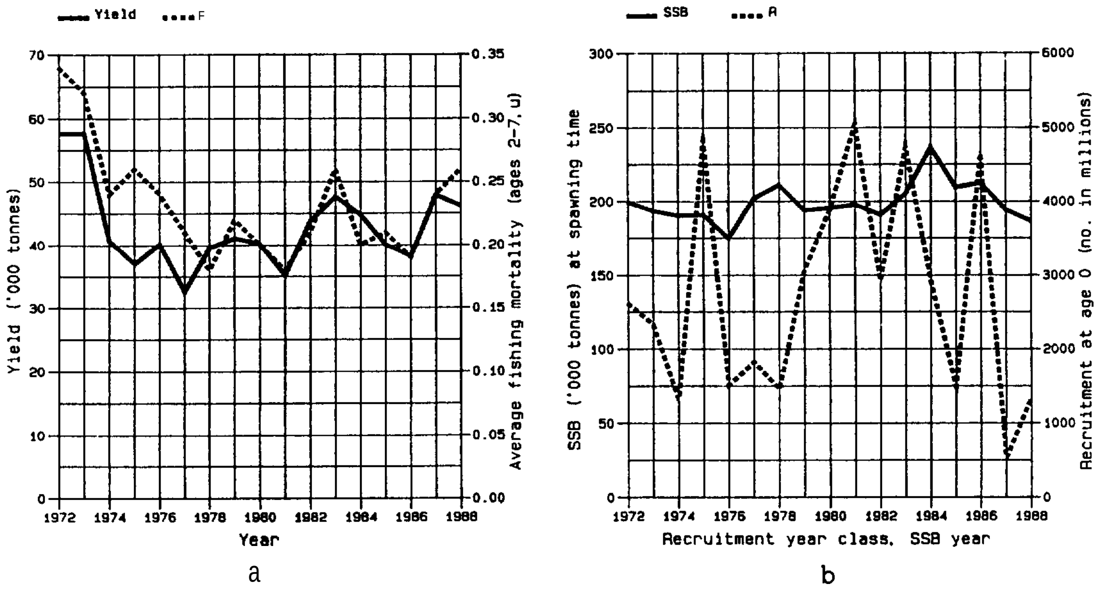

Figure 18. Herring in Sub-division 28 and the southern part of Sub-division 29 (central and northern part of the Eastern **Gotland** Sea) 1972-1988.

a. Trends in yield and fishing mortality (F).

b. Trends in spawning stock biomass (SSB) and recruitment **(R).**

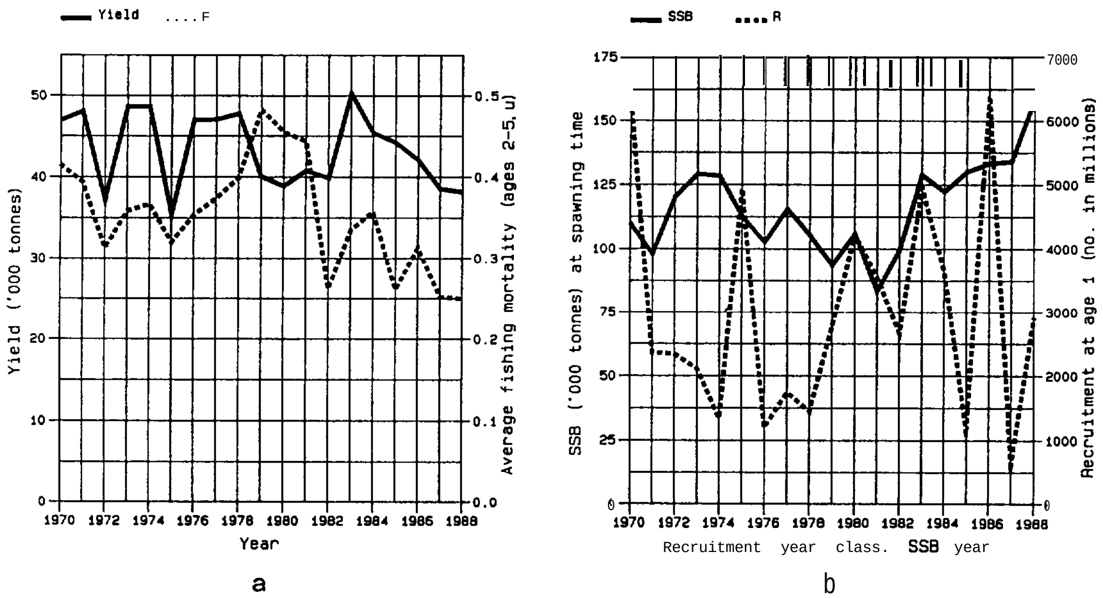

Figure **19.** Herring in Sub-division 32 (Gulf of Finland) 1970-1988.

a. Trends in yield and fishing mortality (F).

b. Trends in spawning stock biomass (SSB) and recruitment **(R).**

Herring in the Gulf of Finland

The catches of herring in the Gulf of Finland (Sub-division 32) reached a maximum value in 1983 and have since declined by 24% (Figure 19a). Fishing mortality has decreased since 1979 and the present level (0.25) is close to $F_{0.1}$ (0.22).

The spawning stock has remained fairly stable at 80,000-130,000 t, but since 1984 it has increased due to the recruitment of two strong year classes (1983 and 1986) and a reduction in fishing mortality (Figure 19b). No trend can be observed in recruitment, but it fluctuates widely.

SUMMARY

A number of factors influence the size of commercial fish stocks in the Baltic Sea area, ranging from the influence of natural conditions to the level of catch in the fishery. While information is lacking on the influence of environmental conditions on the pelagic stocks in the Baltic Sea area, environmental influences on the demersal, i.e., bottom or near-bottom living, species of fish and shellfish can clearly be seen in certain areas.

In the southern Kattegat, an increasing frequency of periods with oxygen deficit in the summer and autumn has particularly affected the stock of Norway lobsters. This has caused a northward movement of the fishery, so that in 1988 and 1989 there was no commercial fishing for Norway lobster in the southern Kattegat. The stress caused by the oxygen deficit in this area is also believed to be linked to an increased prevalence of certain viral diseases in the dab population. As low oxygen values are not found in the spring during the time of the pelagic phase of the eggs and larvae of cod, plaice, dab, and sole, and certainly not at the depths of 0-15 m where the eggs and larvae are found, oxygen depletion has had no effects on the survival of the eggs and larvae of these species in this area.

Similarly, in the Belt Sea and Arkona Sea, the low oxygen concentrations near the bottom in the summer and autumn have restricted the commercial trawl fishery on demersal species in the deeper parts of this area to the period December-April. However, as the spawning time for cod, plaice, and flounder in this area is between February and April, when there is no oxygen deficit, this does not affect the survival of the eggs and larvae of these species.

In the cod spawning areas in the Bornholm Sea, the southern Gotland Deep, and the Gdansk Deep, there is a growing problem owing to the gradually decreasing salinity of the Baltic Sea and the low oxygen concentrations in the bottom waters. Cod eggs require a salinity of 11 or more to keep afloat and an oxygen concentration of at least 2.0 ml/l to survive; thus, the northern border for successful reproduction is found in the Gotland Deep. Successful spawning seems to be positively correlated with the inflow of saline water from the North Sea, which increases the salinity in the bottom layer and improves the oxygen conditions below the halocline. The increasing salinity enables the cod eggs to float higher in the water column, where improved oxygen conditions result in improved survival of the cod eggs.

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7. MICRO-ORGANISMS

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ABSTRACT

During the last years strong efforts have been undertaken to establish a microbiological monitoring in the Baltic Sea area on a routine basis. Regional surveys performed along the middle line of the Baltic Sea during late summer 1982, 1986 and 1988 revealed a remarkable uniformity of bacterial number and activity in the mixed surface layer. In the Kiel Bay between 1986 and 1989 the number of colony forming bacteria showed large interannual fluctuations. At 2 out of 3 stations a decrease of these bacteria was observed. Trends concerning eutrophication by using microbiological parameters, however, are not perceptible due to the still limited number of data.

7.1 INTRODUCTION

During the last two decades the knowledge on bacteria and their role in the cycling of material in aquatic ecosystems has increased substantially. This was mainly attained by a rapid progress in microbiological methodology. Due to the improvement of methods the determination of e.g. the total number of bacteria, their biomass, the rates of destruction of organic substances and production of bacterial biomass has now achieved a high degree of reliability, which nobody would have expected some years ago.

The new results and especially the newly developed concept of the '*microbial loop" have shown that the function of bacteria, or in a broader sense the function of heterotrophic microorganisms, has to be re-evaluated (Hobbie et al., 1977; Azam et al., 1983). Quite a substantial part of the carbon fixed by primary producers is released into the water and then taken up by bacteria. The percentage of organic matter released by the algae may vary considerably according to the environmental and physiological conditions, but it is generally accepted to be in the range of 10 - 50 % (Wolter, 1982; Larsson and Hagström, 1982). These substances are then returned via consumption of bacteria by bacteriovorous animals to the main food chain. Thus, dissolved organic material, released during the process of primary production, but also by prey-predator interactions and by a number of metabolic activities of "higher" organisms, is retained for the use of the higher trophic levels of the food chain, which otherwise would not be able to benefit from organic solutes.

Despite the growing understanding that microbiological investigations are urgently needed to obtain more detailed information about the cycling of matter in ecosystems, large gaps in the knowledge about the role of bacteria in the marine ecosystem still exist. This is due to the general difficulties with microbiological methods and also to the relatively small number of microbiologists working on marine ecological problems. The latter may be the main reason why the microbiological monitoring in the Baltic Sea' area still suffers from the lack of adequate studies.

Routine microbiological monitoring studies in the Baltic Sea area including the parameters total bacterial number, bacterial biomass, number of colony forming bacteria and bacterial production were made only during the last few years. The main areas of such investigations were in the Kiel Bay and eastern Arkona Basin. In 1989 monitoring studies were also initiated along the Polish coastline of the Baltic Proper. In all other areas of the Baltic Sea a routine microbiological monitoring including the above mentioned parameters is not yet undertaken. In some regions similar studies are, however, more or less regularly performed as a part of general microbiological investigations.

Due to the regionally different efforts concerning microbiological monitoring, the short duration of the already existing studies and the generally encountered high degree of temporal and spatial variability of microbiological parameters, it is still premature to draw far reaching conclusions based on the microbiological monitoring about the eutrophication processes in the Baltic Sea.

On the other hand, the initial phase of the microbiological monitoring was characterised by the successful application of a set of microbiological methods in restricted areas in the Baltic. The results will serve as a basis for a more intensive screening of the whole Baltic Sea area.

7.2 REGIONAL DISTRIBUTION OF MICROBIOLOGICAL PARAMETERS

Total bacterial numbers

Three microbiological transects along the middle line of the Baltic Sea were made since 1982. The first (Aug/Sept 1982, Figs. 1-2) covered the whole area between the northern Bothnian Bay and the Kiel Bay. The second was made in Aug 1986 between the Gulf of Finland and the Skagerrak (Figs. 3-4). In July 1988 a third transect was performed roughly along the same route as the second one. The positions of the sampling points are given in Figures 1 and 3. Between 30 and 48 samples were taken at distances of about 20 nm at 5 m depth. The results are shown in Figs. 2, 4, 5 and Table 1.

Table 1. Total bacterial number (TBN: cells x 10^6 ml^{-1}), mean cell volume (MCV: μm^3), bacterial biomass (BBM: μg C l^{-1}) and chlorophyll-a (Chl-a: μg l^{-1}), averaged for different sub-areas of the Baltic Sea. The data were obtained during 3 cruises of RV "Poseidon" (see Figs. 1, 3). Samples were always taken from 5 m depth.

	31 Aug - 5 Sep 1982				22-26 Aug 1986				11-13 and 23-27 July 1988	
	TBN	MCV	BBM	Chl-a	TBN	MCV	BBM	Chl-a	TBN	Chl-a
Bothnian Bay	3.63	0.145	184	1.69						
Bothnian Sea	3.29	0.114	131	1.87						
Gulf of Finland					2.85	0.088	a8	2.49	4.67	1.93
Eastern Gotland Basin	4.01	0.094	132	1.49	3.30	0.099	114	2.14	3.28	2.56
Bornholm Basin	3.66	0.091	117	2.90	3.29	0.107	123	3.24	2.58	2.18
Arkona Basin	3.62	0.114	144	2.40	3.28	0.109	125	3.66	2.93	2.69
Belt Sea	3.99	0.124	173	2.50	3.36	0.106	125	2.64	2.92	2.81
Kattegat					1.15	0.165	66	2.41	1.14	0.97

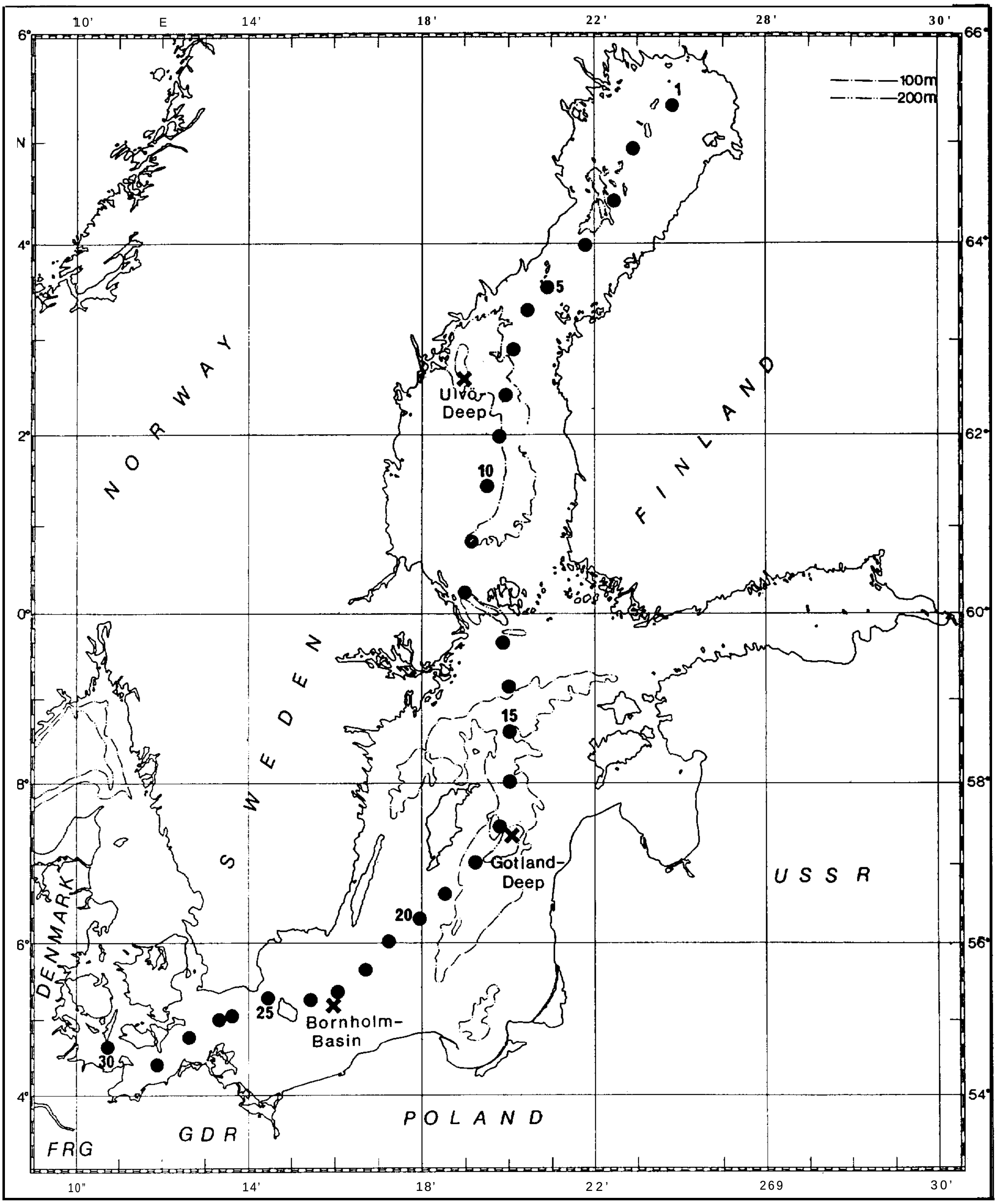

Figure 1. Location of 30 stations sampled in the middle line of the Baltic, 31 August - 5 September 1982.

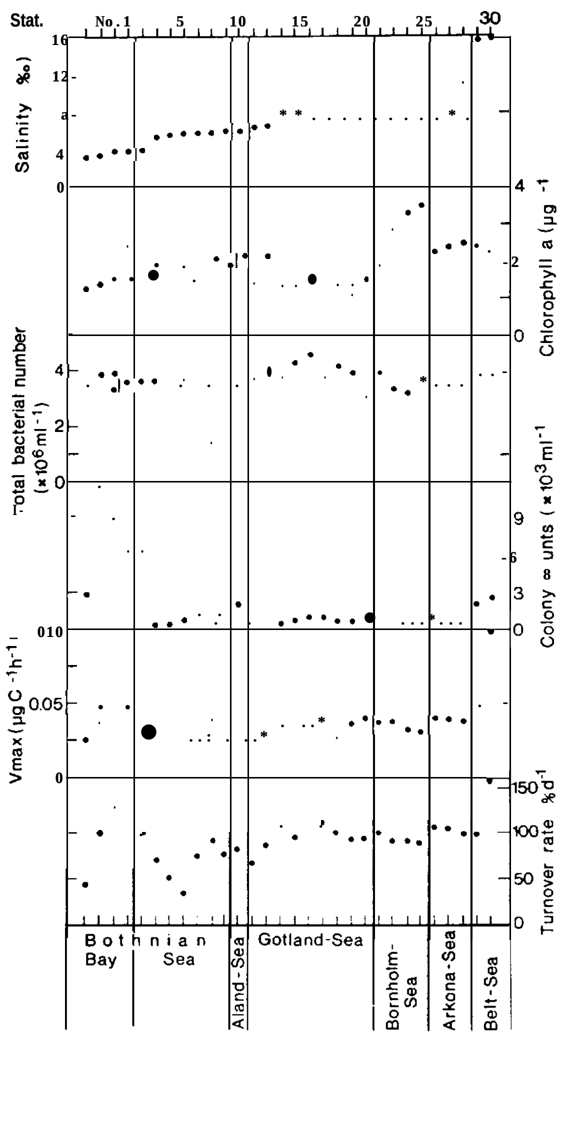

Figure 2. Regional distribution of salinity, chlorophyll-a, total bacterial numbers, colony forming bacteria and maximum uptake velocity of glucose (V_{max}) in 5 m depth along a transect from the Bothnian Bay to the Belt Sea (31 Aug – 5 Sept 1982). The positions of the stations are shown in Figure 1 (Gocke, unpubl).

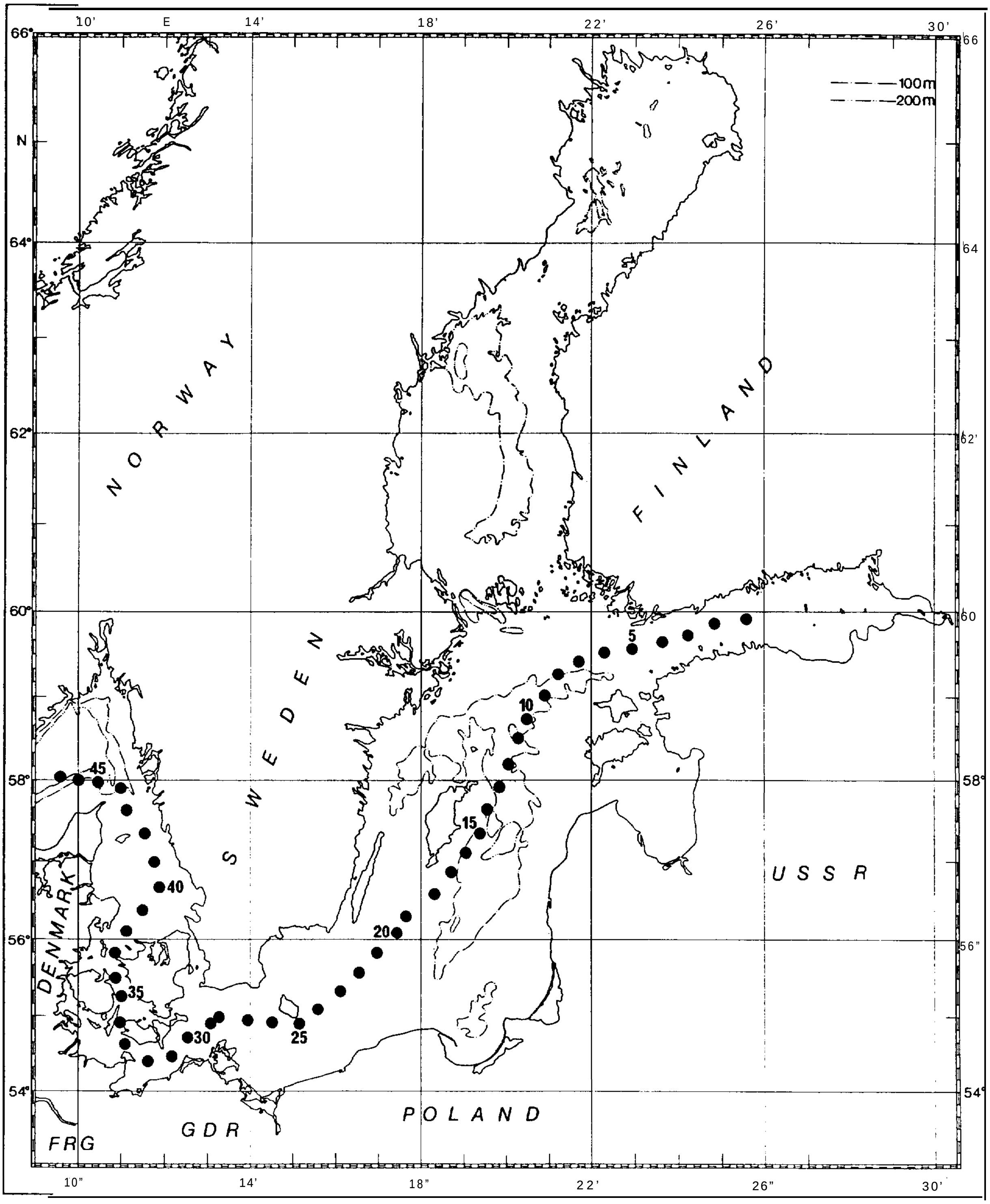

Figure 3. Location of 47 stations sampled in the middle line of the Baltic, 22 August - 26 August 1986. In July 1988 samples were taken along the same transect, the position of the stations, however, may vary slightly.

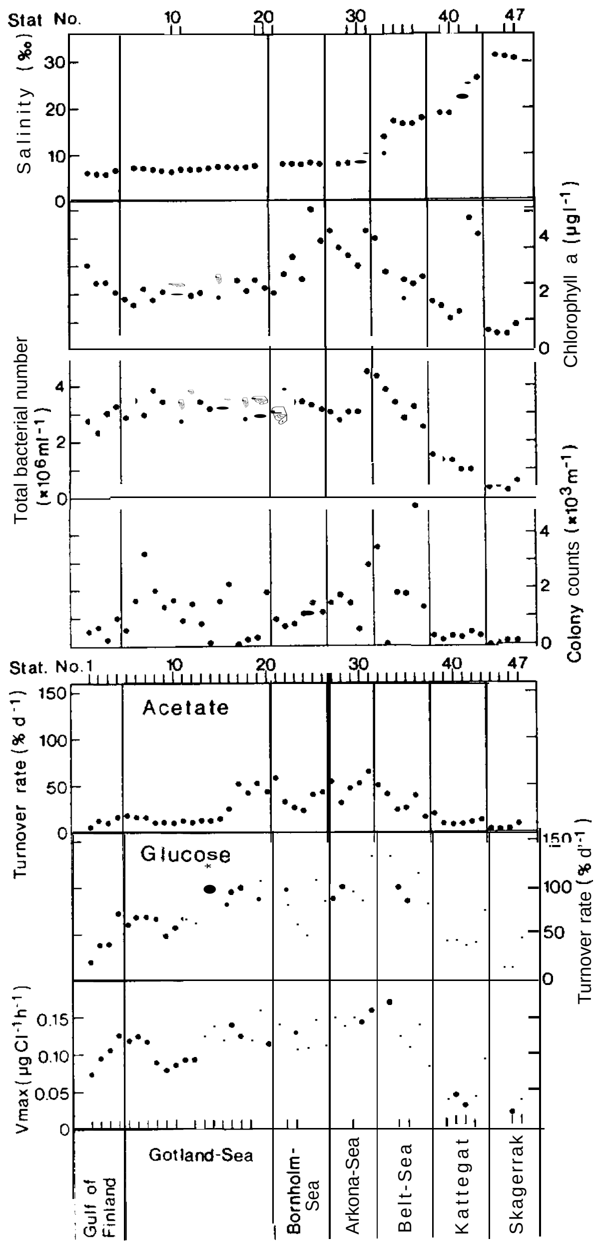

Figure 4. Regional distribution of salinity, chlorophyll-a, total bacterial numbers, colony forming bacteria, turnover rates of acetate and glucose, and maximum uptake velocity of glucose (V_{max}) in 5 m depth on a transect between the Gulf of Finland and the Skagerrak (22 August - 26 August 1986). The positions of the stations are shown in Figure 3 (Gocke, unpubl).

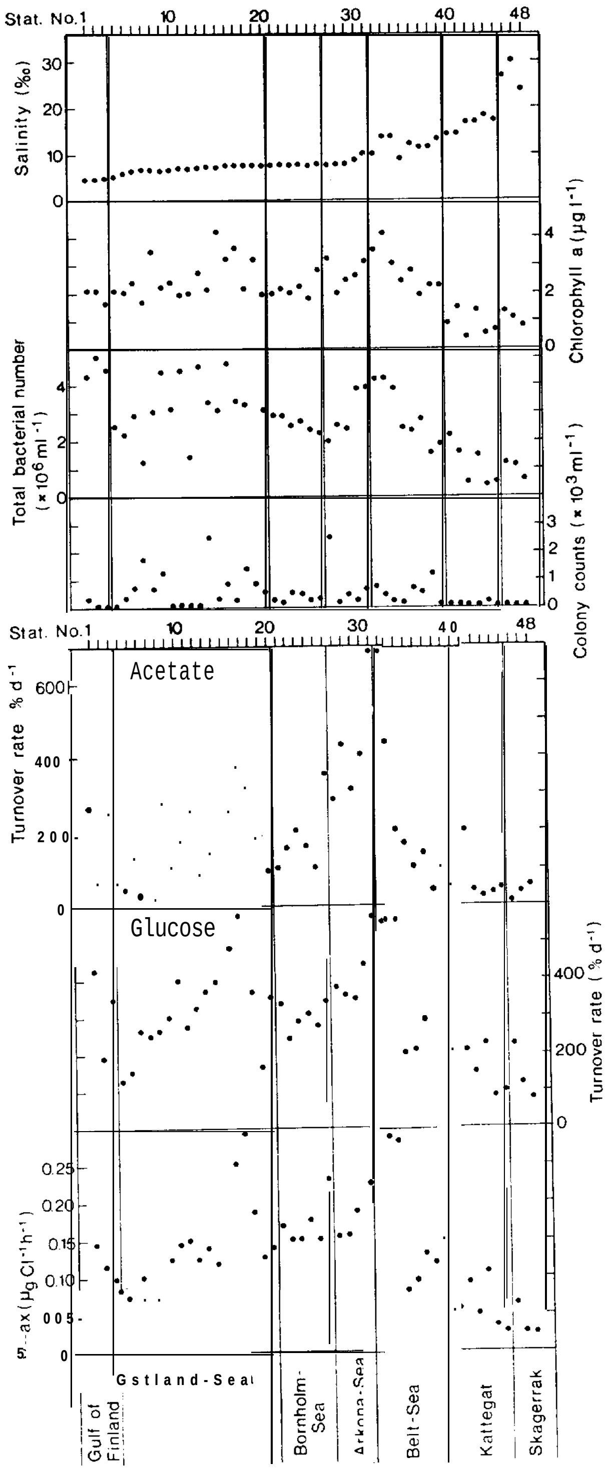

Figure 5. Regional distribution of salinity, chlorophyll-a, total bacterial numbers, colony forming bacteria, turnover rates of acetate and glucose, and maximum uptake velocity of glucose (Vmax) in 5 m depth. Samples between the Kiel Bay and the Skagerrak were taken from 11 to 13 July and between the Gulf of Finland and the Kiel Bay from 23 to 27 July 1988. The location of the transect is similar to that shown in Figure 3 (Gocke, unpubl).

In spite of the substantially differing hydrographical situation within the different areas of the Baltic Sea, the total numbers of bacteria (TBN) show a remarkable regional uniformity (with the exception of the Kattegat and Skagerrak). This holds'true at least for the same cruise, whereas the results between the tree cruises differ somewhat more. Especially in 1982 the low degree of variability was remarkable since TBN values were all (with one exception in the southern part of the Bothnian Sea) between 3 and a little bit more than 4 million cells per ml. In 1986 and 1988 the regional variability was a little bit higher, especially in the Kattegat and Skagerrak the total bacterial number decreased considerably to 1 million cells per ml or less. The latter was also observed by Schmaljohann (1984). Considering the results of the different regions between 1982 and 1988 a slight decrease in total bacterial numbers can be noted. Only in the Gulf of Finland in 1988 considerably more bacteria were found. The reason therefore probably is to be seen in the very heavy blooms of cyanophytes in 1988.

Similar studies along the same transects were not undertaken during the other years. Isolated investigations, however, at a position on or near the transect were performed in 1983 and 1984. The determination of seasonal variability of TBN at the stations BCS III 10 (=BMP K1, southern part of the Eastern Gotland Basin) and BY 5 (=BMP K2, Bornholm Basin) may also serve for comparison. The results do not fit in the above stated picture of the small decline in bacterial numbers between 1982 and 1988. Gast and Gocke (1988) found total bacterial numbers above the thermocline in the Gotland Deep of 5.2 x 10^6 ml^{-1} in 1983 and 5.3 - 6.4 x 10^6 cells ml^{-1} in 1984 (Figs. 6 and 7). At stations K1 and K2 the TBN in surface waters were both 4.8 x 10^6 cells ml^{-1} (data were submitted by Polish microbiologists).

The variations observed between 1982 and 1989 were probably due to the different intensities of cyanophyte blooms and their impact on the bacterial populations (Hoppe, 1981). This can be easily deduced from Fig. 7, which shows that within 6 days the number of bacteria increased from 5 - 6 x 10^6 ml^{-1} to more than 10 x 10^6 cells ml^{-1}. This increase was paralleled by a strong increase in density of cyanophytes. 10 x 10^6 total bacterial number is the highest so far reported for the Central Baltic Proper.

The total number of bacteria in the Baltic Sea area was also analyzed several times by different researchers from the Soviet Union. Pursuing different aims, the authors used different analytical methods, and therefore the results obtained are of different character. However, common tendencies are traced clearly enough when seasonal variations of the total number of bacterial cells are expressed (Tsyban et al., 1985; Andrushaitis et al. 1987). Maximum number of cells (up to 3.8 x 10^6 cells ml^{-1}) was noted during summer. Yurkovskaya (1987) noted that in highly productive areas of the Baltic Proper the total number of bacteria reaches 1 - 4 x 10^6 cells ml^{-1}.

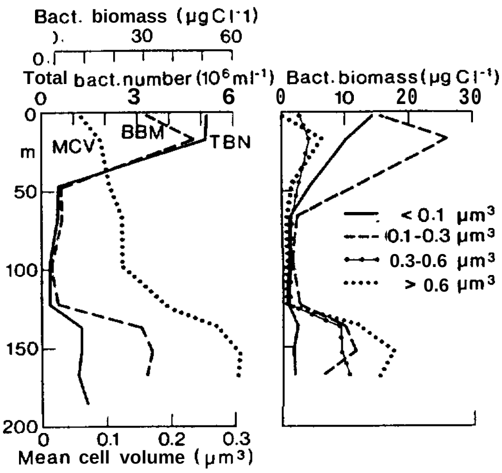

Figure 6. **Vertical distribution of total bacterial** number (TBN), mean cell volume (MCV), total bacterial biomass (BBM) as well as the biomass belonging to different size classes of bacteria. Samples were taken on 20 Aug 1983 in the Eastern **Gotland** Basin **(57°16.5'N, 19°44.7'E)** (Gastand Gocke 1988).

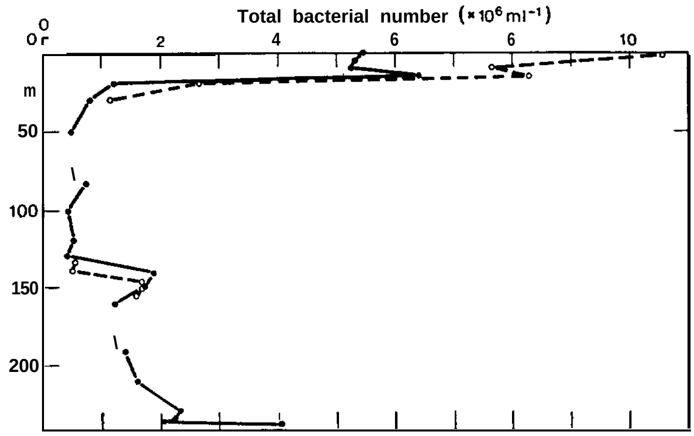

Figure 7. Vertical distribution of total bacterial number in the Eastern **Gotland** Basin **(57°20'N, 20°04'E).** Samples were taken on 1 Aug 1984 (solid line) and on 6 Aug 1984 (broken line) (Gocke unpubl.).

The high bacterial numbers during late summer seem to be typical for the Baltic Proper, whereas the numbers before and at the onset of the spring phytoplankton bloom are very low. A mean value of 0.4 x 10^6 bacteria ml^{-1} was found in April 1975 in the Bornholm Basin (Gocke and Hoppe, 1982), which is only about one tenth of that during summer. Tsyban et al. (1985) and Andrushaitis (1987) found winter values between 0.05 and 1.2 x 10^6 cells ml^{-1}. Nielsen and Richardson (1989) also observed much higher bacterial numbers during May/June than during February/March in the North Sea. During the spring phytoplankton bloom the microbial loop apparently plays a minor role, whereas during the later seasons its importance grows substantially especially when the phytoplankton is dominated by small algae.

It should be mentioned, however, that the total bacterial numbers found in Kiel Bay during summer 1974 were much lower than those observed in the same region in the transect studies mentioned before (Zimmermann, 1977). The same author found a decrease of TBN in winter to about one half of the summer values. Table 2 shows that the relative differences between the early spring and late summer values of total bacterial numbers also were quite small at the stations studied by the Polish microbiologists. The reason for such results may be seen in the extra-ordinary warm spring 1989 which probably caused a very early spring phytoplankton bloom. The situation in the Kiel Bay encountered in 1974, but also in 1988 and 1989, is probably influenced by the vicinity of the surrounding landmasses, which dampens the seasonal oscillations of bacterial number.

Table 2. Seasonal variation of total bacterial number (TBN: cells x 10^6 ml^{-1}) and bacterial biomass (BBM: μg C l^{-1}) at the stations P1 = BMP L1 (Bay of Gdansk), BCS III 10 = BMP K1 (southern part of the Eastern Gotland Basin) and BY 5 = BMP K2 (Bornholm Basin). Values are averaged for the upper 10 m (individual samples were taken from 1 m, 5 m and 10 m water depth).

	Bay of Gdaiisk BMP L1		Eastern Gotland Basin BMP K1		Bornholm Basin BMP K2	
Date 1989	TBN	BBM	TBN	BBM	TBN	BBM
March	4.85	173	3.90	92	3.34	77
May	3.28	80	4.52	119	4.93	102
June	5.05	125	4.03	94	5.07	115
August	3.94	62	4.81	87	4.81	58
September	6.11	149	5.66	150	7.10	169
November	2.56	73	1.94	62	3.52	93

Bacterial biomass

Concerning the size of the bacteria, Table 1 shows the differences between the different regions of the Baltic Sea. The highest mean cell volume (MCV) of the bacteria reaching 0.145 μm^3 was found in the Bothnian Bay, the smallest in the Eastern Gotland and Bornholm Basin (1982), from where it increased somewhat towards the Belt Sea. In 1986 the smallest MCV was observed in the Gulf of Finland and the Eastern Gotland Sea, it then increased towards the west to a small degree. Only in the Kattegat relatively large cells with a cell size of 0.168 μm^3 were found.

Yurkovskaya (1987) noted the seasonal variations of the volumes of bacterial cells: in winter small bacteria (0.20 - 0.24 μm^3) in the water column were found, whereas in summer the volume of bacterial cells in the euphotic zone increases, on average, 2.5 times. These cell volumes are significantly higher than those observed by other investigators, which is probably due to different methods.

The late summer values of bacterial biomass lay between 184 μg C (Bothnian Bay, 1982) and 66 μg C l^{-1} (Kattegat, 1986, see Table 1). The values found at the stations in the Bay of Gdansk (P1 = BMP L1), in the Eastern Gotland Basin (BCS III 10 = BMP K1) and in the Bornholm Basin (BY 5 = BMP K2) (Table 2) for the same season are somewhat lower, which may be due to the different method used to measure the bacterial cell size. Microbial biomass varied seasonally between 7 - 166 μg C l^{-1} in regions studied by Tsyban et al. (1985) and Andrushaitis et al. (1987).

Colony forming bacteria

The number of colony forming bacteria (saprophytes) was remarkably high (up to 10^4 colonies ml^{-1}) only in the Bothnian Bay. In the other subareas values were only around 1 - 2 x 10^3 ml^{-1}. A large scattering in the number of these bacteria was found. In the Belt Sea a certain increase was noticed (Figs. 2, 4, 5). The highest values were found in late summer.

According to Tsyban et al. (1985; 1987) the number of heterotrophic saprophytic bacteria determined by plating on nutrient media amounted to hundreds of cells per ml in the Northern Baltic Proper. An increase in the number to several thousand cells per ml was observed in the direction from north to south.

The vertical distribution of saprophytic bacteria in deeper water layers has, as a rule 2 - 3 maxima: near the surface, in the region of maximum photosynthesis and in the layer of detritus accumulation at a depth of 60 - 80 m, that is at the lower border of the aerobic zone. The hyponeuston layer with the maximum of heterotrophic saprophytic bacteria is located in the surface film. Here the concentration of bacteria is an order of magnitude higher than in the euphotic zone.

As a whole, the distribution of heterotrophic saprophytic bacteria, which assimilate easily oxidizable organic substances, is uneven, especially in the southern part of the Baltic Proper, which corresponds to the complex structure of the water masses.

Table 3. Bacterial production, maximum uptake velocity of glucose (V_{max}) and total bacterial number in the mixed surface layer at different stations in the Baltic Sea area. Data supplied by Finland.

Station	Date 1988	Bacterial production (mg C $m^{-3}d^{-1}$)	Maximum uptake velocity of glucose (μg C $l^{-1}h^{-1}$)	Total bacterial number (x 10^6 ml^{-1})
Kattegat 56°40.06'N, 12°07.00'E	July	1.83	0.011	0.49
Kiel Bay 54°36.00'N, 10°27.02'E	July	13.90	0.180	3.64
Arkona Basin 55°00.00'N, 14°05.02'E	July	19.04	0.125	3.42
Bornholm Basin 55°15.02'N, 15°59.00'E	July	12.85	0.150	4.03
Eastern Gotland Basin 57°20.00'N, 19°51.42'E	July	14.41	0.099	4.83
Entrance Gulf of Finland 59°35.00'N, 23°18.00'E	July	30.36	0.111	7.70

Bacterial production

Measurements of the bacterial production performed more or less simultaneously in several subareas of the Baltic Sea have been made only once up to now. Results are shown in Table 3. The range between the lowest value of 1.83 mg C m^{-3} d^{-1} which was found in the Kattegat, and the highest one of 30.36 mg C m^{-3} d^{-1} observed at the entrance of the Gulf of Finland, is quite high (Heinänen, unpubl.). Bacterial production and total bacterial number were more or less parallel. The high values at the entrance of the Gulf of Finland were obtained during a very intensive bloom of cyanophytes.

Obtained with different methods, the daily bacterial production determined by microbiologists of the Soviet Union varied within a wide range from 20 to 150 mg C m^{-3} d^{-1}; the P/B ratio (the ratio of daily production to biomass) varied from 5 to 40 % and the respiratory activity was 20 µg O_2 day^{-1}.

In the Southern Baltic Proper, the daily production of bacterioneuston and bacterioplankton amounted to 44 mg and 46 mg C m^{-3} day^{-1}, respectively. The calculated mean values of destruction of organic material by bacteria in the Northern, Central and Southern Baltic Proper were 46.7, 36.1 and 36.3 mg C m^{-3} day^{-1}, respectively. In calculations the value of oxygen uptake by one cell at 20°C was used. Assuming the mean value of bacterial destruction in the sea to be approximately equal to 39.7 mg C m^{-3} day^{-1}, it is believed, that the total amount of bacterial destruction in summer amounts to 0. 87 x 10^6 tons of carbon per day (1.74 x 10^6 tons of dry weight).

The ratio between the values of production and destruction (the P/D coefficient) varies for the Baltic Sea from 30 to 42 % per day, which confirms the fact of an excess of destruction processes over the processes of bacterial production (Tsyban and Korsak, 1987).

Although not directly comparable with the bacterial production, the maximum uptake velocity (Vm) and the turnover rates (T_r) of glucose can be taken as a relative indicator for bacterial activity. Table 3 as well as Figures 2, 4 and 5 show high V_{max} in the presence of high bacterial numbers. The relation between the two variables, however, is not as straight as between bacterial production and total bacterial number.

7.3 VERTICAL DISTRIBUTION OF MICROBIOLOGICAL PARAMETERS

The microbiological parameters show high values in the mixed surface layer followed by a sharp decline below the thermocline (Rheinheimer et al. 1989). Near the sediment an increase may be observed in some regions, which depends on the hydrographical situation. Near bottom water movements, which may cause a resuspension of sediments, are probably the main reason for this increase in bacterial number and activity. The vertical distribution is demonstrated for the central part of the Eastern Gotland Basin in Figures 6 - 9. The other regions show a similar distribution, at least, when the surface layer and the deeper water masses are considered.

An exception to the rule of high bacterial numbers and activities in the surface layer and much lower ones in the deeper water was observed in the Kattegat in July 1988. Here the situation was just the other way. The high values of the microbiological parameters below the halocline were most probably due to the massive bloom of ***Chrysochromulina polylepis,*** which occurred during May/June and later settled to the deeper water layers.

A special situation concerning the vertical distribution of bacteria was observed in regions with anoxic water layers. This situation was studied in 1983, 1984, 1986 and 1988 in the Gotland Deep (Figs. 6 - 9). Here the total bacterial number as well as the bacterial activity show a well defined peak around the oxic/anoxic interface with significantly higher values than in the oxic winter water above and in the anoxic water below the halocline. Data suggest (Gocke, unpubl.) that a well defined biocenosis of different bacterial populations and heterotrophic flagellates exists around this interface. H_2S - oxidizing bacteria are an important component of this biocenosis.

Gast and Gocke (1989) report a change in the size-class spectrum of the bacteria, which shifted to larger cells towards the bottom (Fig. 6). The mean cell volume which averaged 0.1 μm^3 in the oxic water of the Gotland Deep, doubled in the oxic/anoxic interface and tripled in the anoxic water. Obviously the bacterial population changed in favour of species more adapted to anoxic conditions. The slowing down of the velocity of metabolic processes due to the lowering of temperature and the scarcity of substrate may also have a certain effect (Seppänen and Voipio, 1971; Meyer-Reil, 1983).

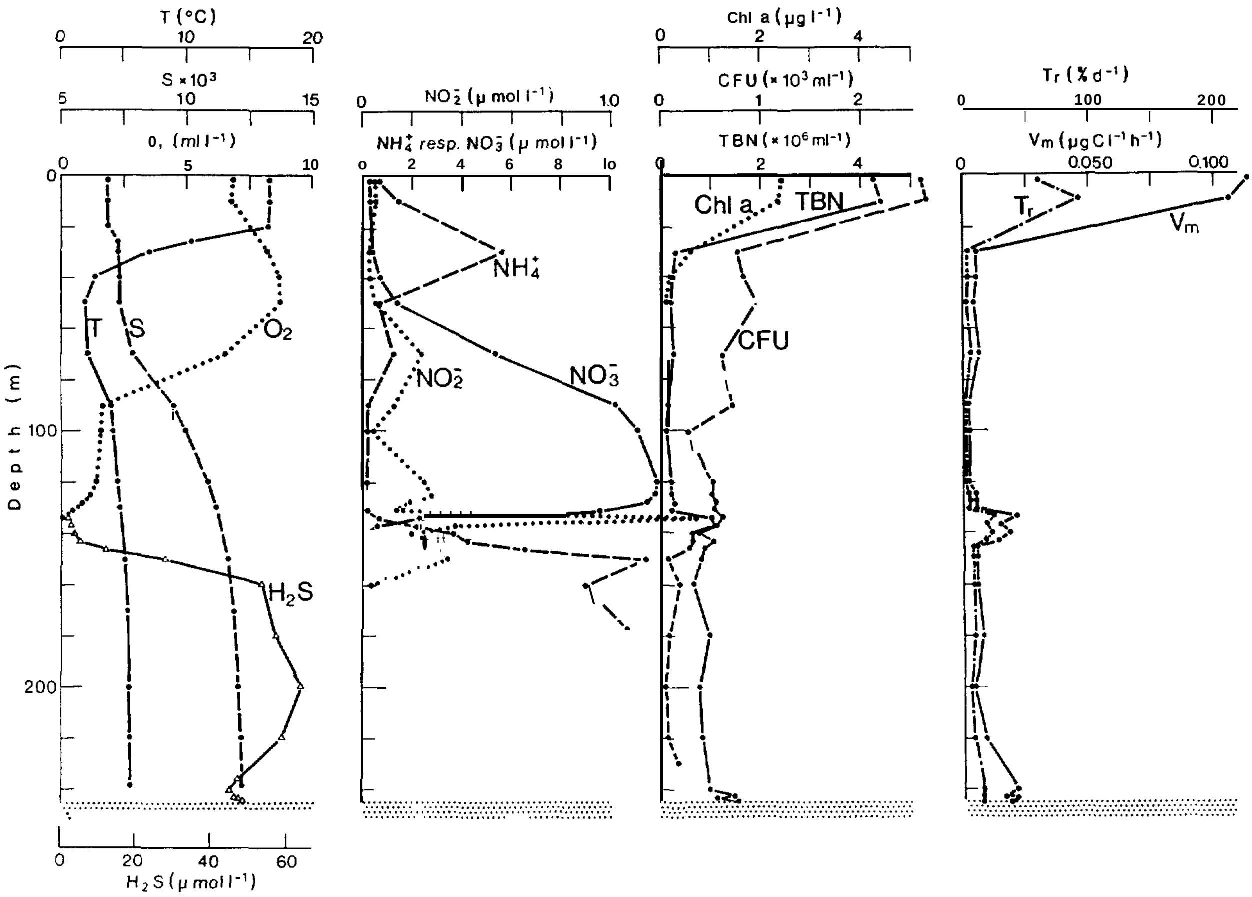

Figure 8. Vertical distribution of temperature, salinity, oxygen, hydrogen sulphide, nitrate, nitrite, ammonium, chloropyll-a, colony forming bacteria (CFU), total bacterial number (TBN), turnover rate of glucose (Tr) and maximum uptake velocity (V_m). Samples were taken from 15 to 17 Aug 1986 in the Eastern Gotland Basin. For location see cross "Gotland Deep" in Fig. 1.

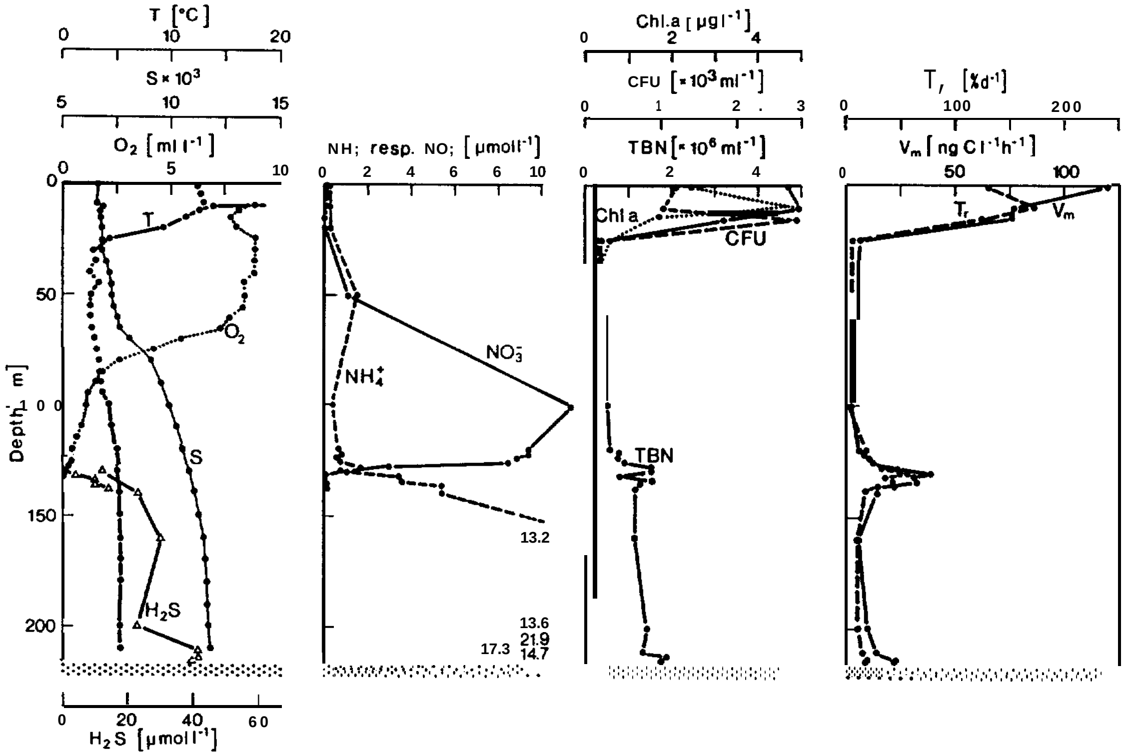

Figure 9. Vertical distribution of temperature, salinity, oxygen, hydrogen sulphide, nitrate, ammonium, chloropyll-a, colony forming bacteria (CFU), total bacterial number (TBN), turnover rate of glucose (T_r) and maximum uptake velocity (V_m). Samples were taken from 19 to 20 July 1988 in the Eastern Gotland Basin (57°20'N, 19°51.4'E).

7.4 ROUTINE INVESTIGATIONS IN THE KIEL BAY

Since October 1985 samples are taken routinely every month at three monitoring stations in the Kiel Bay. The stations are: Boknis-Eck, Kieler Bucht (= BMP N3), and Fehmarn Belt (= BMP N1). Sampling depths are: 2m, 10m and 20m, 15 m at station BMP N3. Beside other parameters, total bacterial number, bacterial biomass, the number of colony forming bacteria, and bacterial production were measured.

Colony forming bacteria

Colony forming bacteria (CFU, saprophytes) are believed to be a good indicator-group for influences of surrounding landmasses. They respond with increased growth to direct organic pollution and to the kind of inorganic pollution which supports and increases algal growth.

Figure 10 shows the number of the saprophytic bacteria from 1986 to the end of 1989 in the mixed surface layer at 2 m depth at station Boknis-Eck. Determination was performed by using the "Plate Count Method" with ZB nutrient agar with a salinity of 8 °/oo. Even though the sampling frequency in 1987 was lower than in the other years, it shows that the average saprophytic numbers decreased steadily. The maximum value of 4 x 10^3 Colony Forming Bacteria per ml in October 1986 was never reached again. The data from 1987 to 1989 display two distinct maxima around May-June and September-October. This indicates the end of the phytoplankton bloom at late spring and autumn.

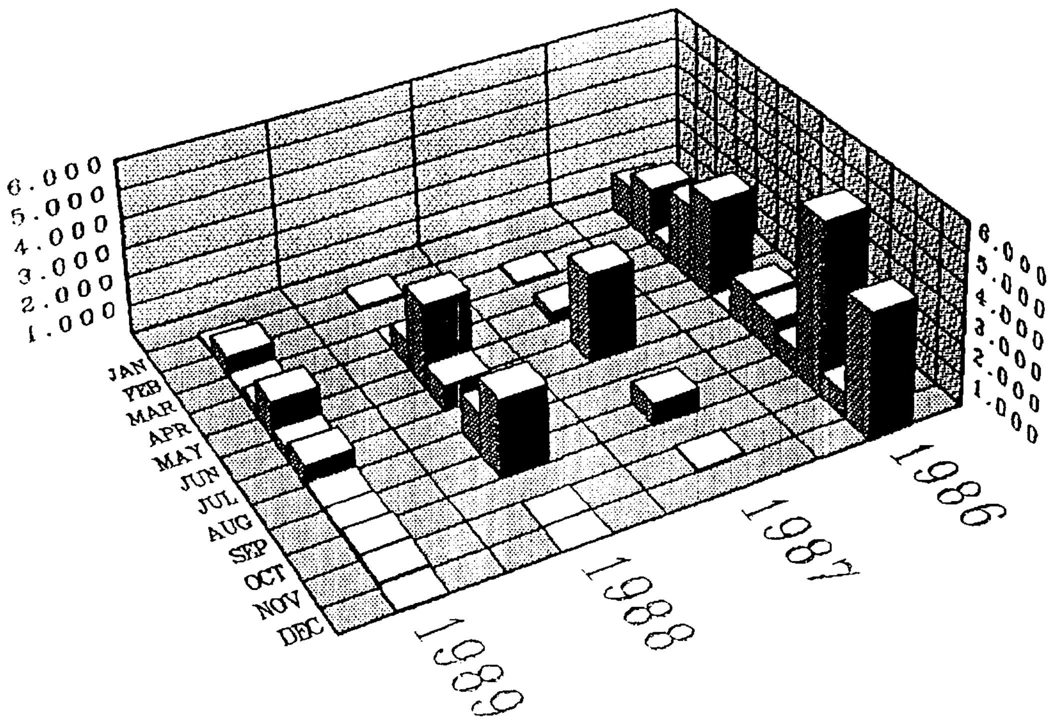

Figure 10. Temporal distribution of saprophytic bacteria (colony forming units / ml) at 2 m depth at station Boknis-Eck (western part of Kiel Bay), 1986-1989.

Figure 11 gives the number of the saprophytes at the station Kieler Bucht (=BMP N3). The general tendency is not as clear as shown for Boknis-Eck. The maximum CFU is less than half the value at Boknis-Eck. The formation of two distinct maxima is not quite evident (except in 1988). The reason might be the position of this station. It is situated in the centre of Kiel Bay at a much greater distance from land than Boknis-Eck, and the water depth is lower (17m), so that resuspension of sediments may occur which may result in irregular changes in saprophytic counts. At station Fehmarn-Belt (=BMP N1) the situation seems similar to Boknis-Eck (Fig. 12).

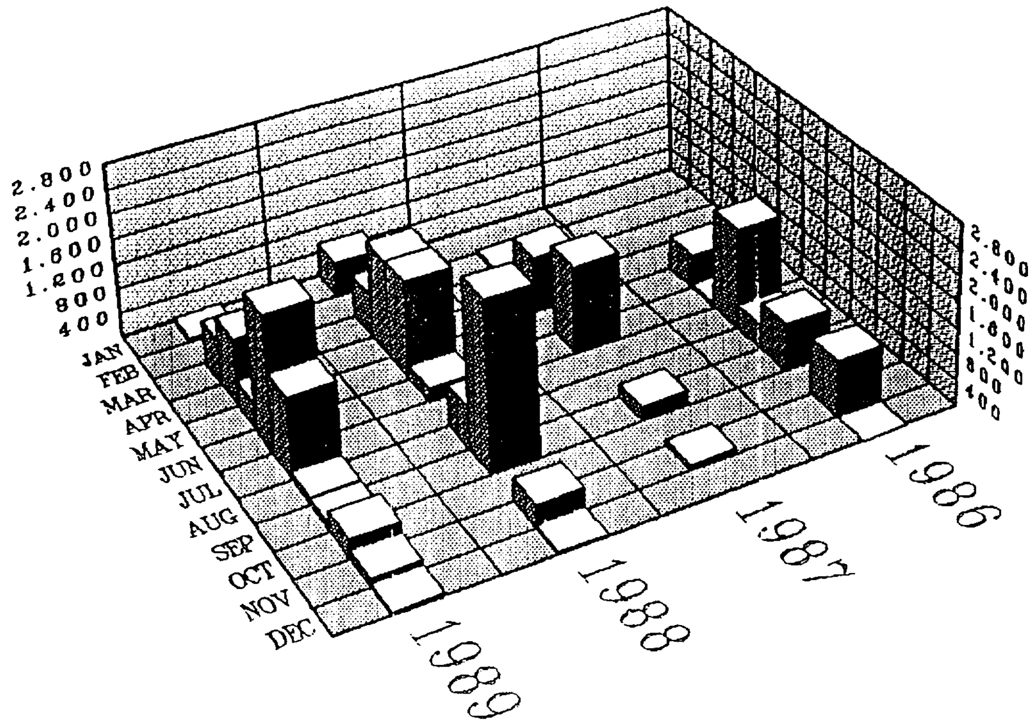

Figure 11. Temporal distribution of saprophytic bacteria (colony forming units / ml) at 2 m depth at station Kieler Bucht (=BMP N3), 1986-1989.

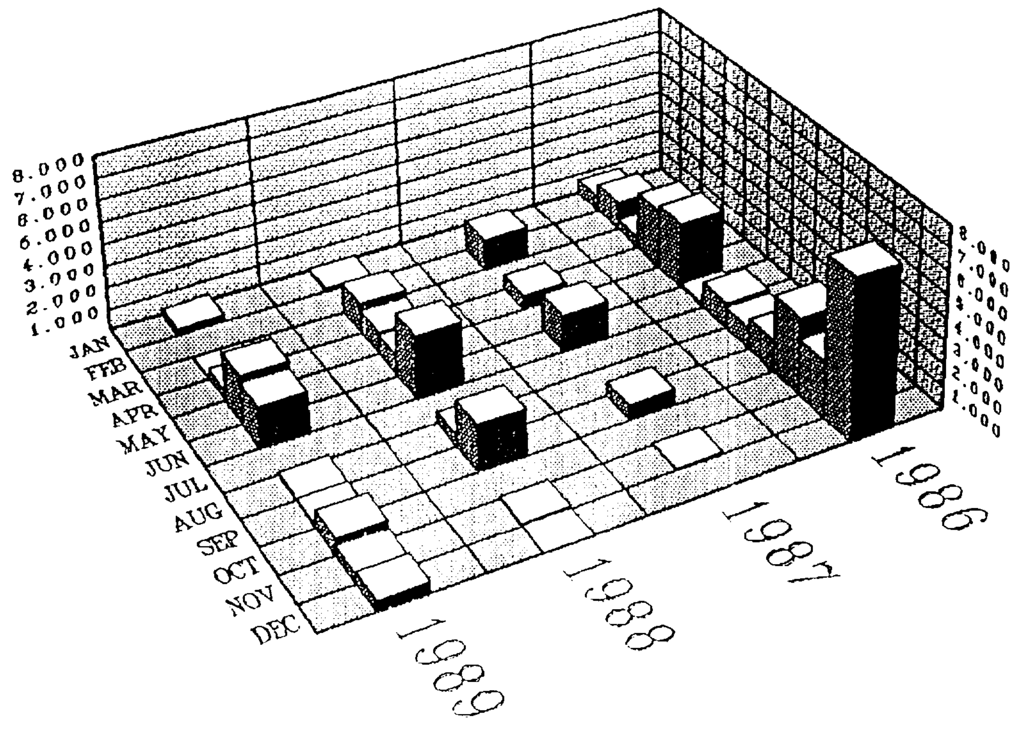

Figure 12. Temporal distribution of saprophytic bacteria (colony forming units / ml) at 2 m depth at station Fehmarn Belt (=BMP N1), 1986-1989.

Bacterial Biomass

The highest level of bacterial biomass amounting to 170 μg C l^{-1} was found during June - July 1988 in the surface layer at station Boknis-Eck (Fig. 13). In 1989 this maximum was observed during the same months. In 1988 a significant difference in biomass between the sampling depths occurred, which was not the case in 1989. One reason might be the more turbulent atmospheric conditions (summer gales) in this year. The minimum of bacterial biomass was found in January 1989. Nevertheless, Figure 13 shows a much higher biomass in the previous January in all depths.

Figures 14 and 15 show comparable data for the other station6 in Kiel Bay. The maxima and minima are at the same time of the year. The biomass in all three sampling depths seems nearly equal. The reason is very probably the same as above mentioned for the saprophytes.

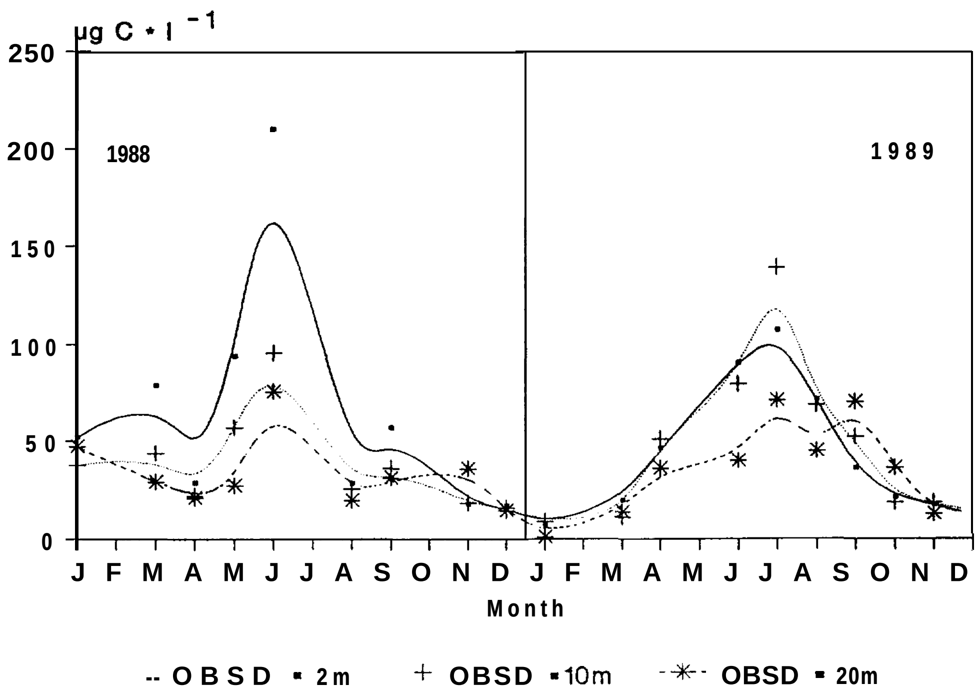

Figure 13. Seasonal variation of bacterial biomass at station Boknis-Eck (western part of Kiel Bay, 1988-1989. OBSD = observed depth.

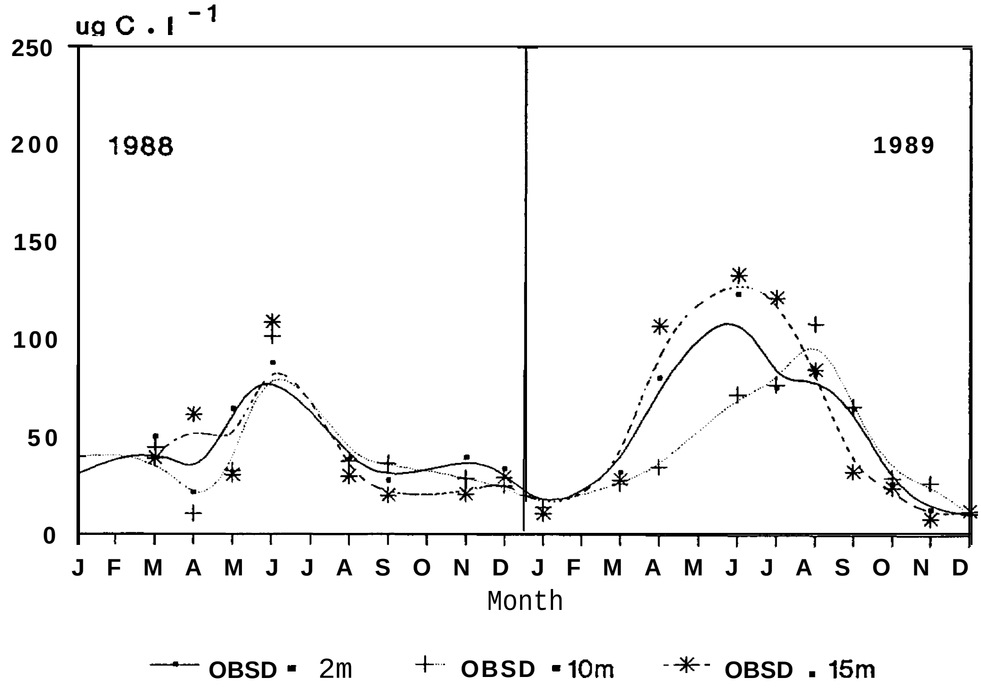

Figure 14. Seasonal variation of bacterial biomass at station Kieler Bucht (= BMP N3), 1988-1989. OBSD = observed depth.

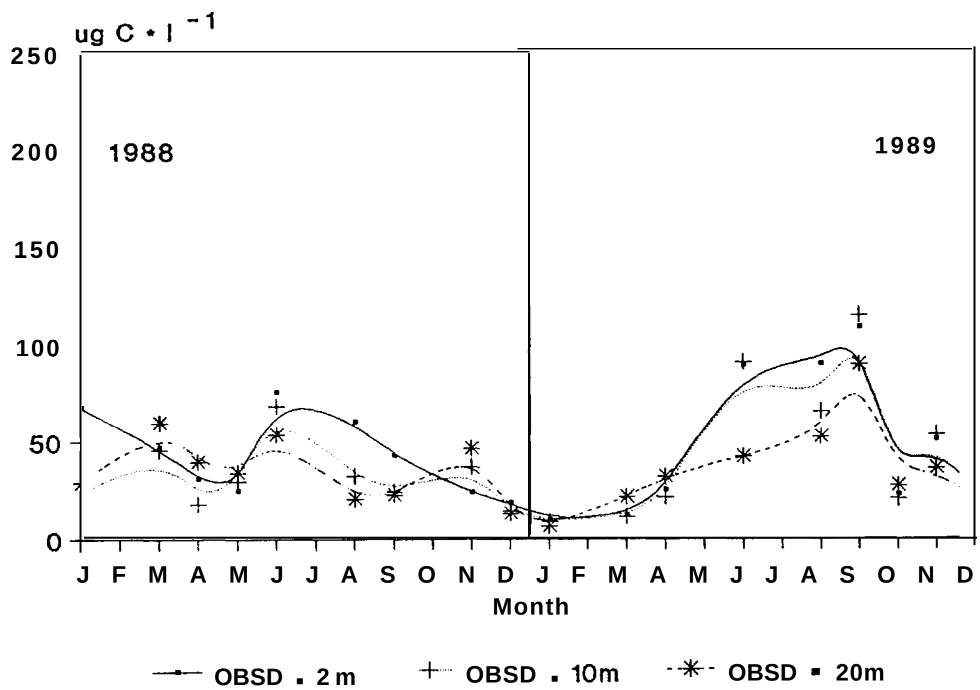

Figure 15. Seasonal variation of bacterial biomass at station Fehmarn Belt (= BMP N1), 1988-1989. OBSD = observed depth.

Bacterial Production

The bacterial production was determined by incorporation of ^{3}H thymidine according to the method of Fuhrman & Azam (1982). This method was introduced to monitoring in May 1988 and after a one year period of testing and adapting the method for routine purposes it is now used on every routine monitoring cruise.

Figure 16 shows the bacterial production in the year 1989 in the mixed surface layer. The lowest production rates can be observed during winter while the highest production occurred in July. The highest value during July was observed at station Boknis-Eck while in December 1989 all three stations showed similar low production rates. A fixed relation between the production rates of the three stations was not recognized.

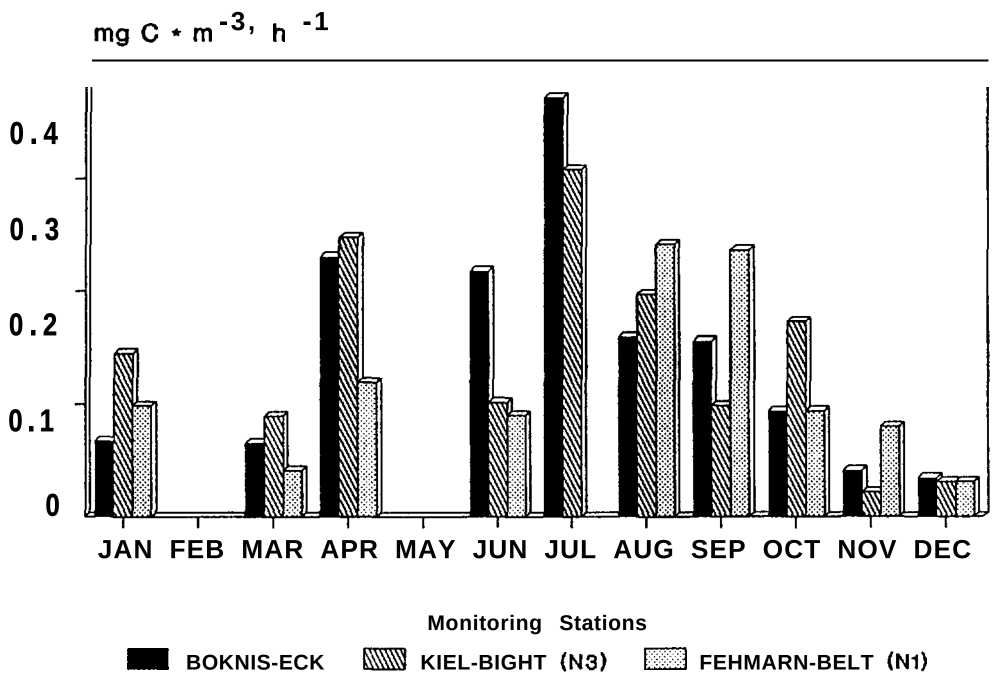

Figure 16. Seasonal distribution of bacterial production (determined by incorporation of ^{3}H thymidine) at 3 Kiel Bay monitoring stations in 1989 (surface layer).

This data are in agreement with the measurements of bacterial standing stock (TBN and bacterial biomass), which exhibit maxima and minima at nearly the same time of the year as shown above. Measuring bacterial production via thymidine incorporation has turned out to be a good additional method to get more detailed information about the dynamics of bacterial activity in marine ecosystems.

7.5 TAXONOMIC CHARACTERISTICS OF THE BACTERIA IN THE BALTIC SEA

The variety of bacterial forms is mainly presented by the following genera: *Micrococcus, Flavobacterium, Microbacterium, Agrobacterium, Citrobacter, Pseudomonas, Mycobacterium, Aerococcus,* and *Bacillus. Micrococcus* and *Pseudomonas were* the most abundant and most frequently observed. In winter the species composition did not practically differ from the forms isolated in summer (Tsyban et al., 1985).

In further investigation (Pfeifere and Platspira, 1986) 50 microbial cultures isolated from the Baltic Sea area were analyzed; 63% proved to be able to transform benzo(a)pyrene (BP); they belonged to the genera *Pseudomonas, Mycobacterium, Bacillus, Arthrobacter, Flavobacterium* and *Micrococcus.*

In coastal waters of the Baltic Sea area BP-oxidizing forms accounted for 0.01 - 22 % of saprophyte counts (Pfeifere, 1987). The variety of the genera of bacteria isolated from littoral areas confirmed the data obtained earlier.

Pseudomonas putida, Pseudomonas. sp., *Bacillus pumilus, B. cirailans, Flavobacterium sp.* and *Micrococcus* sp. prevail in the active silt (Sharma et al., 1987).

7.6 INDICATORY GROUPS OF MICRO-ORGANISMS

One of the most important biological responses to marine pollution consists in the appearance of the so-called indicatory forms of aquatic organisms. Micro-organisms are the most responsive components of marine biocenoses. Their biochemical activity is the basis for the indication of pollutants in the sea (Tsyban, 1978; Tsyban et al., 1985).

Heterotrophic saprophytes, paraffin-oxidizing, PCB-oxidizing and BP-oxidizing forms are conventional indicators of hydrocarbon pollution in the sea, they are adapted forms. The maxima of chemical pollution coincide with the maximum values of the most probable number (MPN) of relevant groups of micro-organisms (Tsyban et al., 1987).

A study of the distribution of micro-organisms oxidizing polychlorinated biphenyls and polycyclic aromatic hydrocarbons was started in 1978 (Tsyban et al., 1985). Since then numerous studies have corroborated the correlation between the presence of hydrocarbon pollutants in the marine environment and hydrocarbon-oxidizing and hydrocarbon-tolerant micro-organisms (Fig. 17).

Chronic pollution of the marine environment by petroleum hydrocarbons resulted in the fact that marine microflora adapted itself to the matching pollutant and acquired an ability to destruct these chemical compounds. It was known earlier (Tsyban et al., 1981) that 50 - 80 % of cultures of heterotrophic saprophytic bacteria isolated from the Baltic Sea water developed actively in the presence of petroleum hydrocarbons which is indicative of their adaptation.

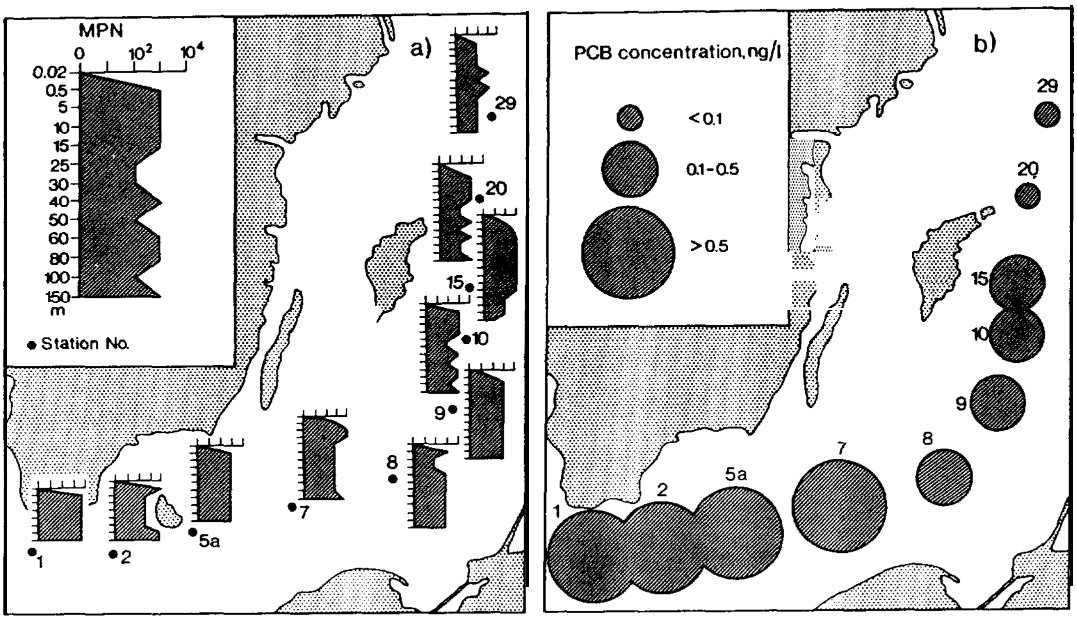

Figure 17. Vertical distribution of the most probably number (MPN, cells/ml) of PCB transforming micro-organisms (a), and PCB concentration in 0.5 m depth (ng/l) (b) in the Baltic Proper in June 1987 (submitted by the Soviet Union).

At the same time saprophytes were investigated, whose metabolism resulted in appearance of intermediate decay products which are often more toxic than the initial substrates. The biomass of microorganisms assimilating these substances is toxic for many aquatic organisms, especially at early stages of their development. It is therefore possible to consider the appearance of abundant indicatory microflora as the secondary marine pollution.

7.7 ENVIRONMENTAL CAPACITY OF TRE BALTIC SEA

The concept of environmental capacity founded theoretically and confirmed by actual materials is described in the work by Izrael and Tsyban (1985).

Micro-organisms are the only component of the ecosystem able to eliminate toxicants under conditions of low temperatures prevailing in the water column of seas in middle latitudes. A decrease in the temperature in winter, however, results in reducing the destruction ability by more than one order of magnitude. Therefore the input of organic pollutants and particularly petroleum hydrocarbons is dangerous at lower temperatures ***since an*** accumulation of xenobiotics in the biota and sediments may result.

Calculation5 of the oxidation rates of hexadecane showed that at relatively low temperature, up to 97.5 $\mu g\ l^{-1}$ of hexadecane could be oxidized by microbial populations per day. It was determined in 1987 that, on average, 15.6 $\mu g\ l^{-1}$, 2.3 $\mu g\ l^{-1}$ and 36.6 $\mu g\ l^{-1}$ of hexadecane were oxidized per day in the Northern, Central and Southern Baltic Proper, respectively. Decomposition rates of hexadecane increase in the direction from north to south-west. As a whole, 24.7 $\mu g\ l^{-1}$ of hexadecane, on average, could be oxidized per day in the pelagic compartment of the Baltic Sea.

It is known that in the presence of oil in water, paraffin and olefin hydrocarbons are subjected to microbial degradation within 1 - 2 days. Aromatic hydrocarbons have more stable molecules (Tsyban et al., 1980). It has been shown by in situ model experiments and consequent chemical analyses that marine micro-organisms in the Baltic Sea area could destruct 5.5 $\mu g\ l^{-1}$ of the aromatic hydrocarbon benzo(a)pyrene within 1-2 days, which accounted for 15 % of its initial concentration. Thus, in the 0-l m layer 1.5 % of benzo(a)pyrene is destructed per day (Izrael and Tsyban, 1985).

Taking into consideration the results of model experiments carried out in 1987 as well as the influx and residence time of the toxicants in the water body, the value of the environmental capacity of the Baltic Sea ecosystem with respect to PCBs and BP were calculated. The environmental capacity with respect to PCBs amounted to 0.0015 - 0.0026 $\mu g\ l^{-1}\ yr^{-1}$ and 32 - 44 t yr^{-1} in relation to the entire Baltic Sea area. The environmental capacity with respect to BP proved to be equal to 0.001 - 0.004 $\mu g\ l^{-1}\ yr^{-1}$ and 30 - 84 t yr^{-1} for the entire Baltic Sea area, respectively.

Comparing the obtained data of the environmental capacity with the current input of PCBs and BP into the Baltic Sea, it is possible to conclude that from the ecological point of view the discharge of the compounds under study does not exceed the permissible values. This conclusion, however, is only justified if the input of anthropogenic pollutants to the Baltic Sea ecosystem would be uniform. If the input of pollutants is uneven, the most hazardous for the ecosystem would be the sources that can produce a local loading in the adjacent area amounting to more than 0.0015 $\mu g\ l^{-1}\ yr^{-1}$ for polychlorinated biphenyls and above 0.001 $\mu g\ l^{-1}\ yr^{-1}$ for benzo(a)pyrene. These values are 5 and 3 times more than the current average inputs of these pollutants to the Baltic Sea.

SUMMARY

When the First Periodic Assessment of the State of the Marine Environment of the Baltic Sea was written, no routine microbiological monitoring programmes yet existed. Today even if the database is still quite limited, the situation has become much better, since at least in some areas of the Baltic Sea routine monitoring is now undertaken.

Regional surveys during the late summer in 1982, 1986, and 1988 covering large areas of the Baltic Sea demonstrated a remarkable uniformity of bacterial number and activity in the mixed surface layer. Only in the Kattegat and Skagerrak these parameters showed significant lower values.

The differences between the years, in which the studies were undertaken were relatively small. Between 1986 and 1989 the microbiological monitoring in the Kiel Bay showed large interannual variations in the number of colony forming bacteria. At two stations (Boknie-Eck and Fehmarn-Belt, N1) a steady decrease of these bacteria could be observed, which, however, was not noted at station Kieler Bucht (N3). But to show a trend concerning the eutrophication by using microbiological parameters is not yet possible due to the limited number of data.

Concerning indicatory groups of bacteria, a clear relation between the character of marine pollution and physiological properties of the microbial populations could be demonstrated. Due to the pollution of some areas by petroleum and chlorinated hydrocarbons, bacterial populations adapted themselves to low concentrations of toxicants and acquired the ability to decompose such compounds as PCBs and PAHs. An increase in the number of indicatory micro-organisms and their activity in the pelagial of the Baltic Sea is indicative of the growing influence of the anthropogenic factor. It does, however, not necessarily indicate the increase in eutrophication of the pelagic ecosystem.

Microbiological studies play still a somewhat minor role in the biological monitoring of the Baltic Sea. Since microbial activity in a general sense is an integrated part of dynamics of organic and inorganic matter pools in the sea, and budgeting these processes is not possible without the knowledge of bacterial production and destruction patterns, it is suggested to maintain the microbiological monitoring on a broader basis in subsequent periods of monitoring the marine environment of the Baltic Sea. This means that the Contracting Parties should increase their effort5 to cover the whole Baltic Sea area with microbiological monitoring studies as frequently as possible.

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8. TRACE ELEMENTS

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ABSTRACT

The following report contains an up-dated knowledge on the different types of suspended particulate matter in relation to their origin, fate, chemical and physical properties.

Trace element data in fish and shellfish, gained from a Baseline Study, from different investigations and from national monitoring programmes provide some insight into geographical differences in the levels of the contaminants involved.

Based on time series, temporal trends of some elements over a period of 8 to 10 years are presented.

8.1 INTRODUCTION

Reliable measurements are a pre-requisite for determining trace elements in various compartments of the marine environment and for understanding the processes which control their transport, distribution and ultimate disposal.

Earlier results cited in the literature must be regarded with great caution since they very often reflect the unawareness of problems in marine trace element chemistry which involve contamination during sampling and analysis and other accuracy deficiencies. The tendency to find unrealistically high trace element levels was a common feature for the majority of data generated in the 1970s and partly in the 1980s.

During recent years, improvement in the understanding of the trace element behaviour in the marine environment has been achieved. This has been made possible because some of the problems associated with systematic errors during sampling and analysis have been overcome. However, intercomparison exercises on trace metal measurements in seawater and biological material conducted under the auspices of the

International Council for the Exploration of the Sea (Bewers et al., 1985, Berman et al., 1988a and 1988b) have stressed the further need for laboratories involved in environmental surveys and monitoring *programmes* to pay much more attention to analytical control measures.

The transport of trace elements through ocean systems depends on a variety of interacting physical, geochemical and biological processes. At present, knowledge is not extensive enough for the identification and exact quantification of the pathways of trace elements in the marine environment. However, some general hypotheses as to the major compartments and fluxes important for trace element transformation and transport may be formulated.

When referring to the occurrence of trace elements in marine waters, it is essential to distinguish between "dissolved" and "particulate" concentrations.

Trace elements must be in a biologically available form to have an impact on different aquatic organisms. The biologically available form of elements is, in many instances, the dissolved form. However, elements associated with particulate matter, both suspended in the water phase.or deposited in sediments, are also available to filter feeders and some other benthic organisms.

The distribution of trace elements both in the "dissolved" and "particulate" form in the marine environment is controlled by their chemical properties, by biological activity and physical mixing and circulation of water masses.

Removal of trace-elements from sea water occurs by one or both of two processes: biological uptake and sedimentation of biogenic material, and/or co-precipitation or adsorption of elements on sinking particulate matter.

Sediment traps have proved to be useful instruments to record the vertical fluxes of settling particles. In particular, the use of radionuclides from the Chernobyl fallout has proved to be an effective method to study processes of adsorption of trace elements onto suspended matter and its flocculation and downward transport from surface water to sediments. Marine plankton blooms play obviously a dominant *role* in these processes. Blooming plankton excretes large amount5 of organic substances, which bind terrigenous suspended matter, including adsorbed trace elements, into macroflocs. The flocs represent a main medium for gravitational settling. Measurements of the flux of Chernobyl radionuclides revealed that these flocs have sinking rates larger than 10 m *per* day (Rempe et al. , 1987).

At the end of the phytoplankton bloom, the feeding activities of, and excretion from, zooplankton induce the aggregation of finer suspended matter to larger particles, so called fecal pellets, which have also high sinking rates thereby transferring trace elements from the mixed layer of the sea to the bottom waters.

At the sea floor, the freshly deposited material is harvested by benthic organisms and/or incorporated into the sediment.

From the mass balance point of view, the depositional flux of trace elements is partly compensated by remobilization of adsorbed elements by diagenetic processes. Furthermore, strong currents or storms may promote an intensive interaction of the water phase with surficial sediments leading to a renewed re-suspension of fine-grained material and returning it to the water column for further transport and exchange processes.

Principally, an assessment of trace elements in the Baltic Sea should be based on a critical evaluation of existing data regarding the compartments water, sediments, suspended particulate matter, and biota.

Kremling had given a comprehensive overview on dissolved trace elements in waters in the background document of the First Periodic Assessment of the State of the Marine Environment of the Baltic Sea (Kremling, 1987). In order to avoid a repetition of facts which were already well documented, it was decided that at present a special section on trace elements in waters of the Baltic Sea was not necessary.

A sub-group on Baltic Sediments was set up under the ICES Working Group on the Baltic Marine Environment with the intention to prepare a critical review of contaminant and geochemical data from sediment studies carried out in the Baltic Sea. Considering these activities and bearing in mind that relevant and sufficient sediment data would be soon available, it was decided to disregard sediments in the context of the "Trace Elements" chapter.

New information have been collected since the above-mentioned first periodic assessment with respect to trace elements in suspended particulate matter and biota. Therefore, in the following treatise emphasis was placed on these two compartments.

8.2 PARTICULATE TRACE METALS (PTM)

L. Brügmann[2]

The distribution of trace metals (TM) in the water column of the Baltic Sea is influenced very much by their interaction with suspended particles. In general, marine particles contain high TM contents (see Table 1). The production, sinking and decomposition of abiotic and biotic particles are now known to be important for controlling the TM distribution within the ocean (Bruland, 1983; Buat-Menard, 1986; Fowler and Knauer, 1986).

Probably, the major sources of particles found in the Baltic Sea are in order of significance (SCOR, 1988):

a) Continental material, mostly silicates, transported via rivers and the atmosphere.

From the riverine supply of particulates which is in order of 7 500 000 t/y (Baltic Marine Environment Protection Commission, 1986) only some per cent are delivered to the open Baltic Sea. The atmospheric particulate load is generated through wind erosion of land and through volcanic eruptions, and is removed to the sea by precipitation-scavenging and by dry deposition.

Table 1. Trace metal content of suspended particles, zooplankton fecal pellets and molts, and marine snow ($\mu g/g$ dry weight) (Savenko, 1988; SCOR, 1988).

Element	Suspended particles	Fecal (fresh)	pellets (sediment traps)	Molts (fresh)	Marine snow (collected in-situ)	Plankton (mixed)
Al	3000	28490	20800-74900		36420	100
SC	0.5	2.8	4-15	0.03		0.2
V	22		59-114			4
Cr	125	38		5.3		10
Mn	140	243	768-2110	12	148	10
Fe	8800	24000	21600-43600	232	12800	800
co	5	3.5	10-15	0.8		1.5
Ni	70	20		6.7	25	10
cu	145	226	308-650	35	10	40
Zn	640	950	< 20	146	40	300
Se	8	6.6		1.9	-	4
Ag	4	2.1		2.9	-	0.4
Sb	5	40	< 5	0.8	-	0.1
Au	0.4		<0.1-0.16		-	
Hg	5	0.34		0.17	-	0.2
Pb	180	34		22	9	20
Th	0.4	4			-	0.1
La	1.2	30	24-49	-	-	0.8
Ba	400	200	192-526	-	-	100
Cd	6	9.6		2.1	3.4	3

b) In-situ production of particles by
primary (autotrophic) particle production,
secondary (heterotrophic) particle production (growth of bacteria, micro- and macro-zooplankton, nekton, particles in the range of 0.1 to 1 000 000 μm),
other biogenic particles including
* fecal matter, particularly tightly packed pellets ("fecal pellets") which can be extremely important as transporters of mass and energy,
* "marine snow" (amorphous aggregates (Alldredge and Silver, 1988)),
* plankton hard parts and carcasses (bio-detritus, primarily from silica and calcium carbonates, from radiolarians, coccoliths, foraminifers, diatoms, crustacean molts, pteropod shells),

production of non-biogenic particles (inorganic precipitates/ Colloids, e.g., formation of Fe/Mn-oxides/hydrous oxides, of barite, pyrites, etc.),
benthic "fluffs" or flocs (un-consolidated, very fluid and structure-less material at the sediment-water interface).

c) Anthropogenic sources of particles, transported to the Baltic Sea both by rivers and through the atmosphere.

d) Human activities causing re-suspension (e.g., by dredging, mining, trawling, boating, mooring grounds, fish farming...).

e) Cosmogenic sources of extraterrestrial material.

Despite this wide variety of sources for particulates, for an assessment of the contamination of the Baltic Sea with trace metals, the PTM level may be used as valuable tool due to the close interaction of natural and anthropogenically generated particles with the general TM load in the surrounding water, atmosphere and in surface sediments. This interaction of dissolved TM with particulates is reflected by processes such as adsorption, co-precipitation, bio-accumulation, re-suspension, exertion, etc.

The published information available regarding the concentrations of particulates and of the PTM in waters of the Baltic Sea had been already reviewed in the "First Periodic Assessment" (Baltic Marine Environment Protection Commission, 1988; Chapter 3.1.2, pp. 97-105). In the meantime, only a few additional published PTM data became available. They shall be considered here:

In September 1981, about 30 samples of particulate matter were sampled from the Bay of Gdaiisk on 0.4 µm polycarbonate Nuclepore filters (Brzezinska-Paudyn et al., 1985) These samples were analyzed on 19 elements by ICP-OES after HNO_3, $HClO_4$ decomposition in a teflon bomb. The main results are given in Table 2.

The filtration efficiency and composition of the collected material was studied using different pore-sizes (nuclepore filters with pore-sizes of 0,2, 0.4, 0.6 and 1.0 µm) and filter types (Nuclepore 0.4 µm, Millipore 0.45 µm, Whatman glass fibre 0.45 µm). The mass-fraction remaining on 1.0 um size filters represented only 27% of the matter collected on 0.2 um filters and about 40% of material yielded with 0.4 µm filters. The ratio between the composition of material collected with 0.4 µm Nuclepore and with 0.45 µm Millipore filters differed significantly from element to element and was in most cases higher than unity (see "0.4/0.45" in Table 2).

In June 1980, from four stations off the Polish coast more than 60 samples of suspended matter collected on acid-cleaned Sartorius cellulose acetate filters (0.45 µm) were analyzed on Cd, Cu, Pb and Zn following decomposition with 6 M HCl/40 % HF in a teflon autoclave (Skwarzec et al., 1988). In Table 3, the results of these investigations are summarized which, however, seem to be relatively high regarding the percentage of particulate Cd in relation to the total concentration. For zinc, the dissolved and particulate concentrations are several times higher than observed by other authors in more recently performed studies.

Table 2. Particulate trace metals (PTM) in the Bay of Gdansk, 1981.

Element	PTM -concentration µg/l	Metal content µg/g	Filter recovery ratio 0.4/0.45
Part.M.	6430		0.7
Ca	50	7950	1.3
Al	47	3500	0.5
Mg	41	6460	1.0
Fe	53	8810	1.7
P	7.0	1310	1.4
Mn	3.0	480	1.1
MO	0.09	15	1.5
Zn	4.0	630	1.1
Ni	0.23	40	1.2
Pb	0.30	52	
cu	0.59	96	1.5
co	0.08	13	1.3
Cr	0.43	100	2.1
Cd	0.08	13	1.6
As	0.38	69	1.6
Se	0.15	25	
Sb	1.35	221	1.6
Ag	0.20	37	1.5

Table 3. Mean concentration of particulate Cd, Cu, Pb and Zn off the Polish coast, 1980; percentage of particulate metal concentration from the total concentration (dissolved plus suspended); metal content of suspensions.

	PTM µg/l	%	µg/g
Cd	0.06	68	34 - 84
cu	0.14	25	56 - 344
Pb	0.18	68	75 - 952
Zn	1.9	15	897 -3203

Studies in 1980/1981 on PTM in waters of the Baltic Sea and parts of the adjacent NE Atlantic (Briigmann, 1986; n=230), were continued during further expeditions in 1983 and 1984. The results obtained on PTM at station "BY 15" (= BMP J1) in the Gotland Basin were discussed particularly with respect to the relation between dissolved and particulate metal fractions in anoxic waters (Briigmann, 1988). In the mixed surface layer, Fe was mainly present in particulate form (0.2-8.5 μg/l). Particulate Pb (3.4-67 ng/l) represented always more than 50% of the total concentrations. The PTM fraction of Ni (l-60 ng/l) was almost negligible whereas in anoxic layers for Zn (10-40 ng/l) and Cd (0.7-22 ng/l) substantial percentages of their total metal concentrations were noticed in particulate form. In 1984, at 200 m depth the suspended matter was extremely rich in Zn (6.4 mg/g) and Cd (51 μg/g). Close to the bottom (235 m), Zn (2.8 mg/g) and Cd contents (13 μg/g) were still relatively high. Here, for particulate Cu (lo-170 ng/l), for which the PTM fraction increased gradually with water depth up to 50% of the total concentration, maximum content (1.5 mg/g) in suspended matter was observed. Mn-rich particles (170 mg/g) were found close to the "redoxcline" at about 150 m, probably due to re-oxidation and precipitation of oxides/hydroxides.

Automated electron probe x-ray microanalysis (EPXMA) was used parallel to bulk AAS determinations to characterise some 15 000 individual particles from 50 samples of suspended matter collected in November/December 1984 from different depths at 18 stations on a transect between the Bothnian Bay and the transient area to the North Sea (Bernard et.al., 1989; Briigmann et.al., 1989). For each particle, 14 minor and major elements were determined and size information data were obtained. Multivariate statistical analyses were invoked to classify the particles in specific types and the abundance variations of these groups were studied. Probably, due to the prevailing winter conditions, it appeared that 80% of all investigated particles contained mostly silicon, and seemed to consist of quartz, K-rich and Fe-rich aluminosilicates. The abundance of $BaSO_4$ particles averaged 5% throughout the Baltic Sea, but amounted up to 44% at some stations. The abundance of Fe-rich particles varied significantly with location and depth and averaged about 4 %. Calcium carbonate particles became more abundant towards the North Sea (see Table 4).

Bulk AAS analyses were carried out on altogether 157 samples. Preliminary results (mean values, standard deviations, ranges) are summarized in Table 5 (in brackets: percentage of the 0.5 N HCl leachable metal fraction).

Table 4. Average composition of the particle types identified by automated EPXMA (normalized to 100 %), with size and shape information.

Group number	Overall abundance (%)	Normalized concentrations (%) SiO_2	Al_2O_3	CaO	Fe_2O_3	MnO_2	TiO_2	MgO	Na_2O	K_2O	Ba	SO_4	PO_4	MAD* (µm)	MID* (µm)	AVD* (µm)	SF*
1	27	86 (7)	4 (5)	1.0 (1.4	1.2 (1.8)	0.8 (1.5)	0.8 (1.8)	0.6 (1.2)	1.0 1.2	1.8 (1.6)	0.7 (1.8)	1.6 1.4)	0.6 (0.7)	1.9 (1.5)	1.0 (0.8)	1.8 (0.9)	1.6 (0.8)
2	37	62 (7)	18 (6)	3 (4)	4 (4)	1.0 (1.8)	1 (2)	1.1 (1.6)	1 (2)	10 (9)	0.7 (.7)	1.8 (2.2)	0.9 (1.·)	2.0 (1.6)	1.0 (0.8)	1.4 (1.1)	1.7 (0.9)
3	16	47 (8)	12 (5)	4 (7)	15 (9)	2 (5)	2 (4)	4 (8)	2.8 (2.6)	5 (5)	1 (3)	3 (4)	1.4 (2.9)	2.1 (1.7)	1.0 (0.8)	1 (1)	1.8 (0.9)
4	5	10 (7)	2 (2)	1.1 (2.4	3 (6)	1.5 (2.6)	0.05 (0.45)	1.5 (1.8)	1.5 (1.8)	1.0 (2.2)	30 (12)	38 (8)	1 (2	1.5 (1.2)	0.9 (0.7)	1.2 (0.9)	1.2 (0.8)
5	4	15 (8)	3 (8)	2 (3)	49 (18)	7 (10)	1.4 (2.6)	2.6 (1.6)	3.5 (2 2)	1.6 (2.2)	1.4 (2.7)	3 (5)	1 (7)	1 (1)	0.6 (0.5)	0.8 (0.7)	2 (1)
6	5	18 (7)	3 (2)	2 (8)	10 (7)	51 (14)	1 (2)	2.6 (1.6)	6 (8)	1.8 (2.7)	1.1 (2.7)	4 (4)	3 (3)	1.8 (1.2)	0.9 (0.6)	1.2 (0.8)	1.8 (0.7)
7	2	10 (7	8 (2)	72 (13	1.7 (2.2)	1.0 (1.4	1.0 (1.8)	2.8 (3.3)	0.9 (0.7)	0.9 (1.6)	1 (2)	1.8 (8.8)	5 (7)	1.8 (1.6)	0.8 (0.7)	1.1 (0.9)	1.7 (0.9)
8	2	15 (11	4 (4)	10 (11)	4 (7)	4 (7)	4 (7)	6 (7)	7 (5)	9 (11)	4 (7)	1 (13)	20 (17)	1.1 (1.2)	0.5 (0.6)	0.7 (0.8)	2.1 (1.1)
9	0.8	15 (10)	4.1 (2.5	2 (4)	7 (10)	1 (2)	58 (17)	2 (1)	1.0 (0.5)	1.8 (1.8)	5 (6)	23 (4)	1.0 (1.8)	1.1 (1.8)	0.6 (0.6)	0.8 (0.8)	1.4 (0.4)
10	0.1	12 (14)	50 (18)	8 (3)	2 (3)	2 (4)	4 (4)	0.5 (0.8	0.7 (1.8)	1.2 (1.9)	0 (0)	4 (6)	22 (18)	1.5 (1.2)	0.8 (0.7)	1.1 (0.9)	1.4 (0.5)

( ) : standard deviation
MAD : maximum diameter
MID : minimum diameter
AVD : average diameter
SF : shape factor *

*) shape factor: ratio between both axis of a particle in an idealized rounded shape (factor 1 = globe)

Table 5. Elemental composition of suspensions from the Baltic Sea, 1984.

Sample specific.	Susp. mg/l	Al %	Ca %	Mn %	Fe %	Zn µg/g	Ba µg/g
Microlayer (.25-.45mm) n=17	.9±.5 (.3-2)	2.8f1.2 (.6-5.2) (17%)	.56±.45 (.09-2.2) (-)	.16±.19 (.03-.85) (100%)	2.2f2.1 (.4-9.9) (55%)	480±320 (230-1540) (95 %)	1410+-910 (560-3750) (61%)
Surface (0.2-10m) n=64	.4±.2 (.1-1.2)	4.2f2.2 (.7-12.9) (17%)	.8±.6 (.12-2.63) (75%)	.7±2 (.05-12) (71%)	2.5il.4 (.4-7.9) (68%)	320±130 (150-790) (94%)	1100+-600 (30-3460) (52%)
Intermediate (20-400m) n=38	.4±.2 (.01-.9)	6.1i9.6 (.8-6.2) (20%)	.8±.6 (.1-2.4) (75%)	1.4f2.6 (.02-14.7) (93%)	3.i3.8 (.4-24.5) (63%)	600±1100 (130-6400) (100%)	920+-600 (70-2940) (47%)
Bottom 10-550m n=38	.7±.5 (.1-2.6)	4.6f2.3 (.4-8.8) (20%)	.8±.7 (.l-3.8) (75%)	1.3±4. (.0l-19.6) (100%)	2.8f1.2 (.5-6) (71%)	400±500 (130-2790) (100%)	960±670 (70-3220) (44%)

Concluding remarks:

1) **There exist furtheron an urgent need for intercomparison and standardization with respect to sampling of particulates and sample pre-treatment (type of filter, filtration technique, volume of filtered water/load per filter area, drying procedure, reduction of contamination risks in the iaboratory ..).**

2) In **general, pressure filtration through 0.4 µm filters of the Nuclepore** type has become more and more the accepted sampling procedure. This is due to experiences in expert's laboratories and is one of the consequences of a questionnaire action within ICES. More than 60 laboratories responded and the majority of them seem to prefer that technique. The surprisingly good results of an ICES Pilot Intercomparison Exercise on TM contents of suspended particulate matter (pre-loaded 0.4 µm/47 mm Nuclepore filters were distributed among 7 experienced/ selected laboratories) reflected that in principle the necessary contamination-free and reproducible digestion techniques and instrumental methods exist for accurate and precise PTM determinations. The most sensitive part of the whole analytical procedure is the sampling.

3) With respect to the high variability in the Baltic Sea ecosystem (strong gradients of background parameters in horizontal and vertical direction, anoxic conditions in central basins, seasonal variations, high organic load...) the pool of data on the concentration of PTM in different water masses and seasons, of the metal content and of the nature (origin, mineralogical and grain-size composition) of the particulate matter is by far not sufficient as reliable information regarding the general contaminant load of the Baltic Sea with PTM.

4) Gradients and differences, observed in concentrations of PTM and of the suspensions and the distribution patterns of metal contents of the particulate8 may always reflect both, anthropogenic influences and natural processes causing re-distribution and generation of suspended particulate matter of different composition and properties.

8.3 TRACE ELEMENTS IN BIOTA

U. Harms'

8.3.1 Introduction

Among trace elements, certain metals (mercury, cadmium, lead, copper, zinc) have received particular interest in ecotoxicology as well as priority and significance in the context of existing convention5 for the prevention of marine pollution. These "inorganic contaminants" have several environmental features in common. They occur, a5 in all other environmental compartments, ubiquitously also in the marine environment (GESAMP, 1976). They are nearly the only class of contaminants which are biologically nondegradable but undergo biogeochemical cycles and have the tendency to accumulate in organisms from different trophic levels of the marine food web (Wood et al., 1968, 1974; Nriagu 1978; Windom and Kendall, 1979; Hutzinger, 1980).

Trace element concentration5 are determined in organ5 and tissues of bioindicators because they provide a time-integrated measure of the bioavailable portion in the respective environment under study. The actual trace element concentrations in selected organisms can undergo considerable fluctuations, mainly as a result of a combination of factors such as seasons, food supply, growth and reproductive cycles. Fish liver is preferably used in the context of monitoring trace elements in biological tissue. Knowledge on the binding and accumulation of metals in liver is far from being complete. Phillips (1980) has written an extensive review qualifying the problems related to monitoring (inorganic) contaminants in the organisms and stressed the importance of measuring lipids in the liver. Grimås et al. (1985) showed that the accumulation of zinc, copper, lead and cadmium by cod (*Gadus morhua)* liver was significantly influenced by the fat content of the tissue analysed. These observations indicated, as the authors concluded, that metals in liver were connected to specific sites and molecule5 in the protein fraction, which varied considerably in dependence of the liver's lipid content. The fact that several harmful trace elements are found at lower concentrations in skeletal muscle of marine and freshwater fish than in inner organs is regarded as indicative of regulatory metabolic exclusion mechanisms, which prevent incorporation or at least enrichment of such elements in muscle tissue (Chow et al. 1974, Bryan, 1976).

In order to understand the extent to which aquatic biotopes have been influenced by man's activities, it becomes crucial to determine contaminant levels in different aquatic compartments including biota with appropriate analytical certainty. Trace analyses are subject to numerous difficultiesincludinginterferences, inappropriate instrumental settings and uncontrolled gain or loss factors, which can complicate measurements and evaluation and introduce biasses in the final results. Accuracy is very difficult to achieve in environmental analyses; yet it is extremely important. The problems of data quality with special reference to the measurement of trace metals in marine samples have been discussed by Topping (1986).

The greatest problem in trace element analysis is the fact that analytical data become more unreliable the lower the concentrations to be determined are. The picture of inherent errors or of only limited reliability is always encountered in analysis whenever very low concentrations in the sub-mg/kg region are to be measured.

As a result of the application of more reliable sampling techniques and analytical procedures knowledge of the actual concentration levels of inorganic contaminants in various compartments including biota has increased. This promoted a better understanding of the behaviour and distribution of such substances in the marine environment.

A primary target of the contributions presented in the following was to compare contaminant levels in different fish species and molluscs from different locations/areas of the Baltic Sea, and to investigate whether pronounced regional differences existed in the contamination levels of the bio-indicators selected. A further question of fundamental interest was to compare data sets from different years and to investigate whether temporal fluctuations and/or trends of the contaminant levels could be identified.

Acknowledgement: The author wishes to express his sincere thanks to the International Council for the Exploration of the Sea for provision of data from the "Baltic component" of the 1985 Baseline Study of Contaminants in Fish and Shellfish. The author received active advice by Mr. Anders Bignert, Swedish Museum of Natural History, on the graphical presentation and evaluation of data reported by Sweden and by Denmark in the framework of their national monitoring programmes. This is gratefully acknowledged.

8.3.2 Geographical Baseline Study 1985

The following chapter contains excerpts from a report on the results of 1985 Baseline Study of Contaminants in Fish and Shellfish (ICES, 1988), which covered the North Atlantic, the North Sea and the Baltic Sea. Areas of North and Baltic Sea sampled in the 1985 Baseline Study are shown in Figure 1. As regards extracting the "Baltic component" from the 1985 Baseline study, primary consideration was given to the two teat organisms herring *(Clupea harengus)* and cod *(Gadus morhua)* which were specifically selected for the Baltic Monitoring Programme (BMP) for the Second Stage. As a complement to information about contaminant levels in fish, blue mussel *(Mytilus edulis)* as representative of non-migrating benthic organisms was included in the investigations.

The selection of metallic contaminants was restricted to those which were obligatory determinanda in the context of the BMP. These were: copper, zinc, cadmium, lead and mercury.

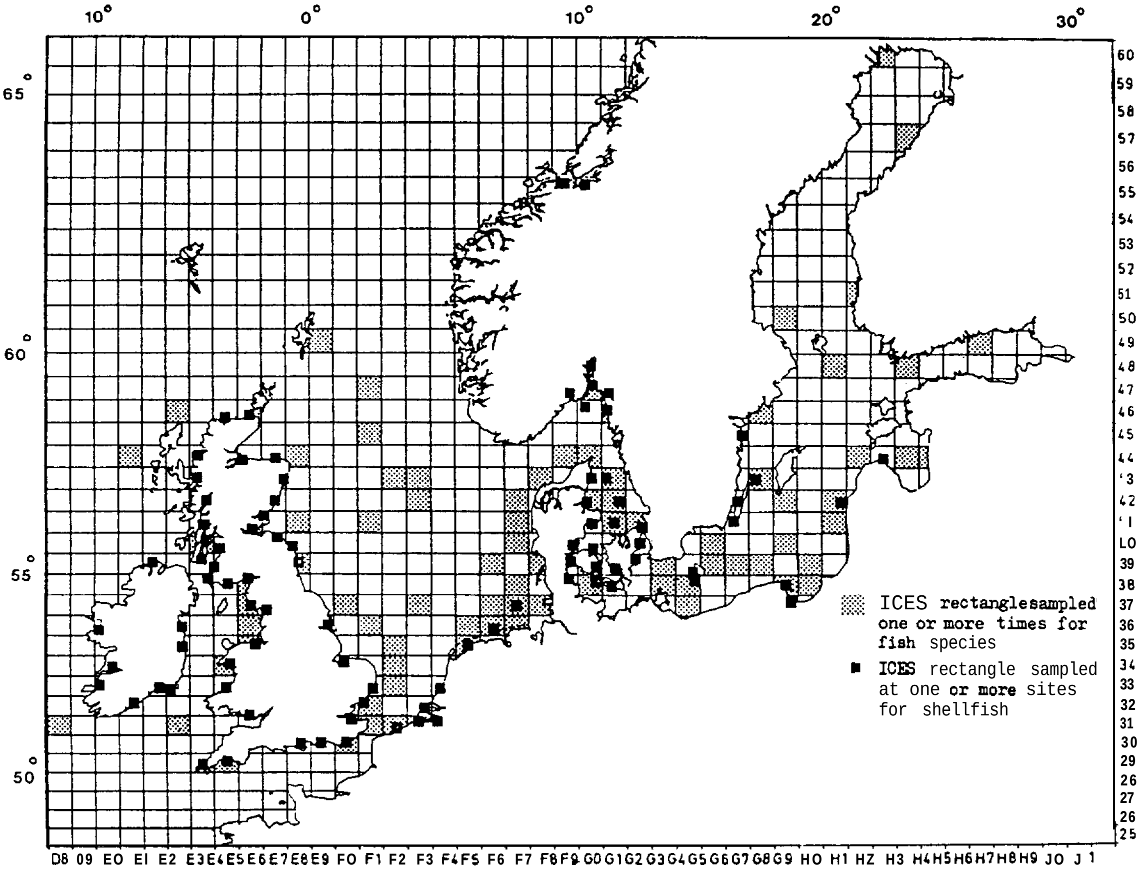


Figure 1. Marine areas in northern Europe sampled in the 1985 Baseline Study of Contaminants in Fish and Shellfish.

The following parameters are summarized in Tables 6-10 for different contaminant/species/tissue combinations:
The minimum (min), the lower quartile (25 %), median (50 %), upper quartile (75 %) and maximum (max) contaminant levels. no is the total number of samples collected for each species, and (n) the number of samples actually used in the assessment after exclusion of unacceptable data.

The criteria used for rejection were derived from the results of the ICES Seventh Intercalibration on Trace Metals in Biological Tissue (Berman et al. , 1988). The Tables contain also the ICES Statistical Rectangles (sampling areas) together with the reporting country for the areas in which concentrations above the upper quartile were observed. In this context, the following statement is quoted from ICES (1988; page 15): "It should be noted that the delineation of quartiles is a relative measure, inherent to the population of data under consideration. Thus, the upper quartile has in itself no implication in an "absolute" sense .

8.3.3 Assessment of data from the 1985 Baseline Study

a) Species-specific differences

Some elements seem to be mutually correlated in fish. As portrayed in Figures 2-3, these conditions become most obvious for copper and zinc in cod and herring muscle tissue. It is also clear from these figures that herring shows a tendency to higher values both of copper and zinc than cod. Conversely, the mercury concentrations are higher in cod than in herring (Fig.4). A possible explanation for these observations may be found in the feeding habits (herring feeding on plankton, cod feeding on invertebrates) of both species investigated.

Mercury concentrations in blue mussels are, with few exceptions, significantly lower in comparison with mercury in fish analyzed in this study (Fig.5). The data indicate "typical" levels for molluscs (GESAMP, 1986).

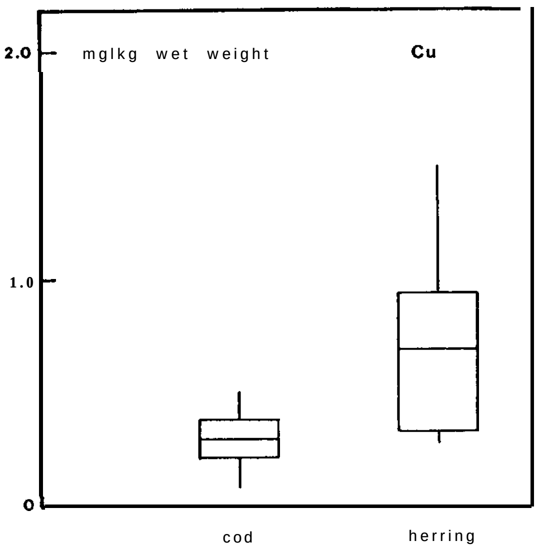
As mentioned earlier, metals are found at lowest concentrations in the muscle of fish and this in itself may be indicative of a regulatory process present in muscle tissue. The obvious exception is mercury (as regards the five elements discussed here), which commonly occurs in muscle tissue of fish at concentrations approaching or even exceeding those in other organs.

It is also worth noting in this context that in view of extreme low levels both of lead and cadmium encountered in fish muscle tissue, and having in mind the difficulties associated with analyses in the extreme trace range it was recommended in the guidelines for the 1985 Baseline Study that these metals were to be analyzed in the livers of fish only. Cadmium and lead data in fish muscle tissue although submitted by some Contracting Countries, were therefore not taken into account in the context of this report.

b) Regional differences

On the basis of findings outlined in Tables 6-10, the level of a metallic contaminant in an area was identified as being "higher" in comparison to the other areas investigated, if at least one of the following conditions was fulfilled:

- **More** than one species/tissue was found with contaminant concentrations above upper quartile.
- More than one sample of the same species/tissue was found to contain contaminant concentrations in the upper category.
- One or several adjacent Statistical Rectangles show a coincidence of upper concentration data.



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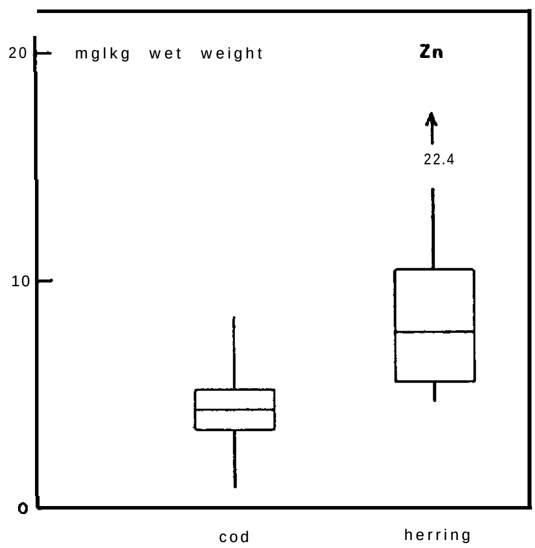

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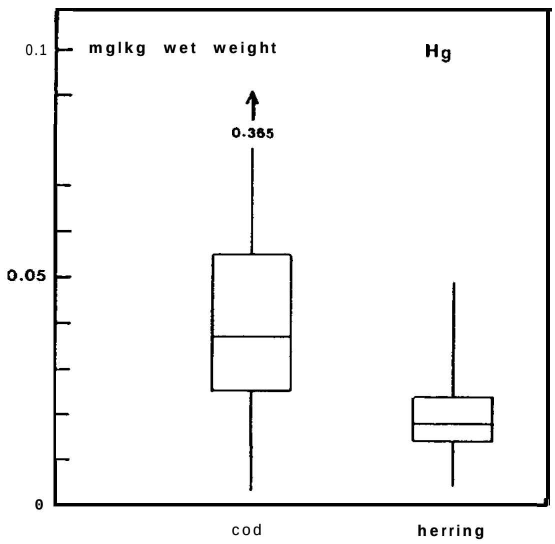
Figure 2. Box-and-whisker plots for copper concentrations (mg/kg wet wt.) in muscle tissue of cod and herring.

Figure 3. Box-and-whisker plots for zinc concentrations (mg/kg wet wt.) in muscle tissue of cod and herring.



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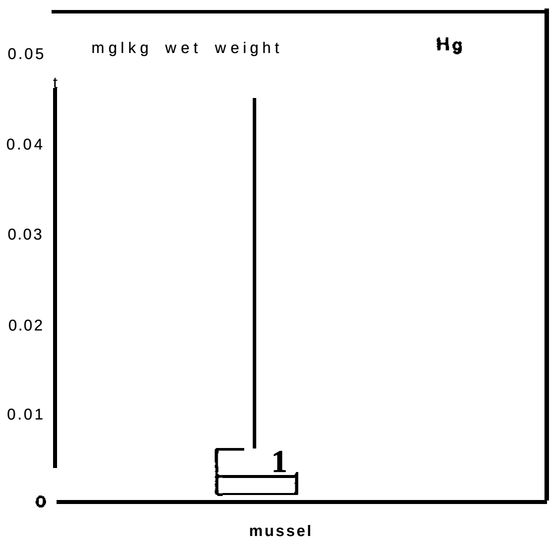

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Figure 4. Box-and-whisker plots for mercury concentrations (mg/kg wet wt.) in muscle tissue of cod and herring.

Figure 5. Box-and-whisker plots for mercury concentrations (mg/kg wet wt.) in blue mussels (whole soft body).

Species	Tissue	n_0	n	min.	25%	50%	75%	max.	Areas with values above upper quartile.
Cod	Liver	20	16	4.81	5.86	7.40	9.22	22.40	PL 3863, PL 40G5, S 42G8, S 43G7, SF 4883
Cod	Muscle	56	54	0.077	0.213	0.292	0.379	0.506	PL 38G4, PL 41H0, PL 40G8, PL 3803, PL 40G5 PL 3969, PL 3967
Herring	Liver	21	16	1.62	4.61	4.96	5.23	6.93	PL 3863, PL 39G9, PL 40G5, S 43G1
Herring	Muscle	35	29	0.26	0.328	0.69	0.94	1.50	PL 3966, PL 39G9, PL 3863, PL 40G5
Blue mussel	Soft body	68	61	0.35	1.03	1.17	1.36	69.30	DK 39G0, DK 38F9, DK 40F9, DK 41G0, DK 42G0 DK 43G1, DK 40G2, DK 38G0, PL 38G8, SU 42H0 SU 4482

Table 6. Results of 1985 Baseline Study of Contaminants in Fish and Shellfish (Baltic part only). Copper concentrations in fish (mg/kg wet weight) and shellfish (mg/kg wet weight).

Species	Tissue	n_0	n	min.	25%	50%	75%	max.	Areas with values above upper quartile.
Cod	Liver	19	7	0.024	0.030	0.036	0.047	0.062	S 43G1, SF 4883
Cod	Muscle	56	2						
Herring	Liver	21	19	0.048	0.088	0.34	0.605	1.50	PL 3863, PL 39G9, PL 3869
Herring	Muscle	33	11	0.010	0.013	0.016	0.049	0.15	SF 5783, SF 51H1, SF 48H0
Blue mussel	Soft body	68	57	0.086	0.156	0.204	0.305	4.61	DK 40F9, DK 42G0, DK 43G1, DK 42G1, DK 41G2 DK 40G2, DR 3962, DK 3964, DK 3864, SU 42H0 SU 4432

Table 7. Results of 1985 Baseline Study of Contaminants in Fish and Shellfish (Baltic part only). Lead concentrations in fish (mg/kg wet weight) and shellfish (mg/kg wet weight).

Species	Tissue	n_0	n	min.	25%	50%	75%	max.	Areas with values above upper quartile.
Cod	Liver	11	11	0.006	0.008	0.012	0.018	0.064	PL 38G3, PL 3969, SU 42H0
Cod	Muscle	41	40	0.002	0.025	0.037	0.055	0.365	DDR38G0, PL 41H0, PL 40G8, PL 39G7, PL 3869 S 43G1, S 43G7, SU 42H0
Herring	Liver	10	10	0.012	0.020	0.027	0.032	0.163	PL 40G5, PL 3863, SU 44H3
Herring	Muscle	36	35	0.004	0.014	0.018	0.024	0.049	D 39G0, D 3863, DDR38G3, PL 3869, S 60H2 S 4667, SF 4986, SU 44H4, S 50G8
Blue mussel	Soft body	58	58	<0.001	0.001	0.003	0.006	0.045	DK 41G2, DK 40G2, DK 38G0, DK 40G0, DK 3962 DK 39G4, DK 38G4, DK 3868, SU 42H0, SU 4482

Table 8. **Results of 1985 Baseline Study of Contaminants in Fish and Shellfish (Baltic part only). Mercury concentrations in fish (mg/kg wet weight) and shellfish (mg/kg wet weight).**

Species	Tissue	n_0	n	min.	25%	50%	75%	max	Areas with values above upper quartile.
Cod	Liver	20	15	0.014	0.027	0.036	0.050	0.105	PL 40G5, PL 38G3, S 4268, S 43G1, S 43G7
Cod	Muscle	56	54	<0.002	0.004	0.006	0.026	0.056	PL 37G4, PL 38G4, PL 40G5, PL 3969, PL 3863 PL 38G8, PL 39G7, PL 3869
Herring	Liver	21	19	0.169	0.346	0.472	0.553	0.663	PL 3863, PL 3969, PL 40G5, PL 3869
Herring	Muscle	33	31	0.001	0.002	0.006	0.034	0.060	PL 3869, PL 3863, PL 40G5, PL 3969
Blue mussel	Soft body	68	61	0.041	0.079	0.100	0.132	1.80	DK 43G1, DK 42G1, DK 41G2, DK 40G2, DK 38G0 DK 38G1, DK 39G2, DK 3964, DK 38G4, DK 3868 DK 3768, SU 42H0, SU 44H2

Table 3. **Results of 1985 Baseline Study of Contaminants in Fish and Shellfish (Baltic part only). Cadmium concentrations in fish (mg/kg wet weight) and shellfish (mg/kg wet weight).**

Species	Tissue	n_o	n	min.	25%	50%	75%	max.	Areas with values above upper quartile.
Cod	Liver	20	18	8.06	13.20	14.60	18.00	33.40	PL 40G5, PL 3863, PL 40G5, S 42G8, S 43G1
Cod	Muscle	58	15	0.800	3.42	4.30	5.22	8.57	PL 38G3, PL 3969, PL 3863, SU 44H1
Herring	Liver	21	17	20.40	25.10	26.80	28.00	38.80	PL 3863, PL 40G5, S 46G7
Herring	Muscle	35	24	4.57	5.58	7.78	10.58	22.40	D 38G3, PL 3863, PL 39G9, PL 40G5, SU 4483 SU 4484
Blue mussel	Soft body	68	57	10.60	13.50	14.60	16.00	29.20	DK 39F9, DK 39G1, DK 38F9, DK 40F9, DK 41G0 DK 42G0, DK 40G2, DK 38G0, DK 39G2, DK 39G4 SU 42H0, SU 4482

Table 10. Results of 1985 Baseline Study of Contaminants in Fish and Shellfish (Baltic part only). Zinc concentrations in fish (mg/kg wet weight) and shellfish (mg/kg wet weight).

Note:

Only data from the HELCOM area have been reported here. The latitude line 57°30' has been used to define the northern extent of the Kattegat.

Data from the Swedish laboratory UCKS were reported on a dry weight basis. The dry weight % was not included and therefore these values could not be converted to a wet weight basis.

< values have been reported by some laboratories, as can be seen in the raw data tables. Although < values will have an effect on the quartile values, they were nonetheless utilised in calculating quartiles.

Quartile values have been rounded. The areas listed in the table with values above the upper quartile are those in which one or more concentrations have been reported above the unrounded 75% (upper quartile) value calculated for the entire data set.

In the raw data, values identified with a * were originally rejected from the baseline study and therefore not included here.

Table 11. Compilation of areas where "higher" concentrations of trace elements were found in biota in the 1985 Baseline Study of Contaminants in Fish and Shellfish.

Sub-Region	Contaminant/area				
	Zn	Cu	Cd	Pb	Hg
Gulf of Riga	44H2,44H3 4404				44H2,44H3, 4404
Eastern Gotland Sea	39G9,44H1	38G8,39G9	37G8,38G8 38G9,39G9	38G9,39G9	38G8,38G9,39G9 41H0,42H0
Bornholm Sea	40G5	39G6,39G7, 40G5	40G5		
Arkona Sea	38G3,39G2 39G4	38G3,38G4	38G3,39G2	38G3,38G4 39G2,39G4	38G3,38G4,39G2
The Sound	40G2		40G2,41G2	40G2	40G2,41G2
Kattegat	41G0,42G0 43G1	42G0,43G1	42G1,43G1	41G2,42G0 42G1,43G1	
Belt Sea	38F9,.39E9 40F9,38G0	38F9,.38G0 39G0 40F9,41G0	38G0,38G1		38G0,39G0 40G0

As outlined in Table 11, the majority of sub-regions identified by the aforementioned procedure lay in the Southern Baltic Proper and the Belt Sea and Arkona Sea, respectively.

It must be stressed, however, that these conclusions are based solely on a limited number of observations unevenly distributed over the Helsinki Convention area. This indicates that countries participating in the study were selective in the way they had chosen sampling locations. As a consequence of the uneven arrangement of sampling, the picture gained through the 1985 Baseline Study remains fragmentary, giving only incomplete information on the true geographical distribution pattern of the trace metals investigated in biota in the Baltic Sea as a whole. From the foregoing discussion on analytical uncertainties in trace element analyses, the intercomparability of data sets is also questionable, and it is not clear whether the "regional" differences observed reflect more the different analytical capabilities of institutions participating in the exercises rather than the actual geographical differences of contamination.

Having this in mind, the results of the baseline study provide only a self-contained data set and Table 11 must not be considered independently of the preceding text.

8.3.4 Synopsis on prevailing levels and temporal trends of trace elements in Baltic fish and shellfish

The most comprehensive information on metallic trace element levels in Baltic herring was reported by Sweden in the framework of its National Environmental Monitoring Programme (Odsjö et al., 1987, 1988). Data sets covering partly a period of 10 years are available for areas 60H2 (Bothnian Bay), 5068 (Bothnian Sea), 4667 (Northern Baltic Proper), 4065 (Southern Baltic Proper), and 4361 (Kattegat).

Extensive time series on trace metal levels in flounder were also reported by Denmark for two locations (in the Sound and the Great Belt) in the Danish marine *environment* (Pedersen, 1989).

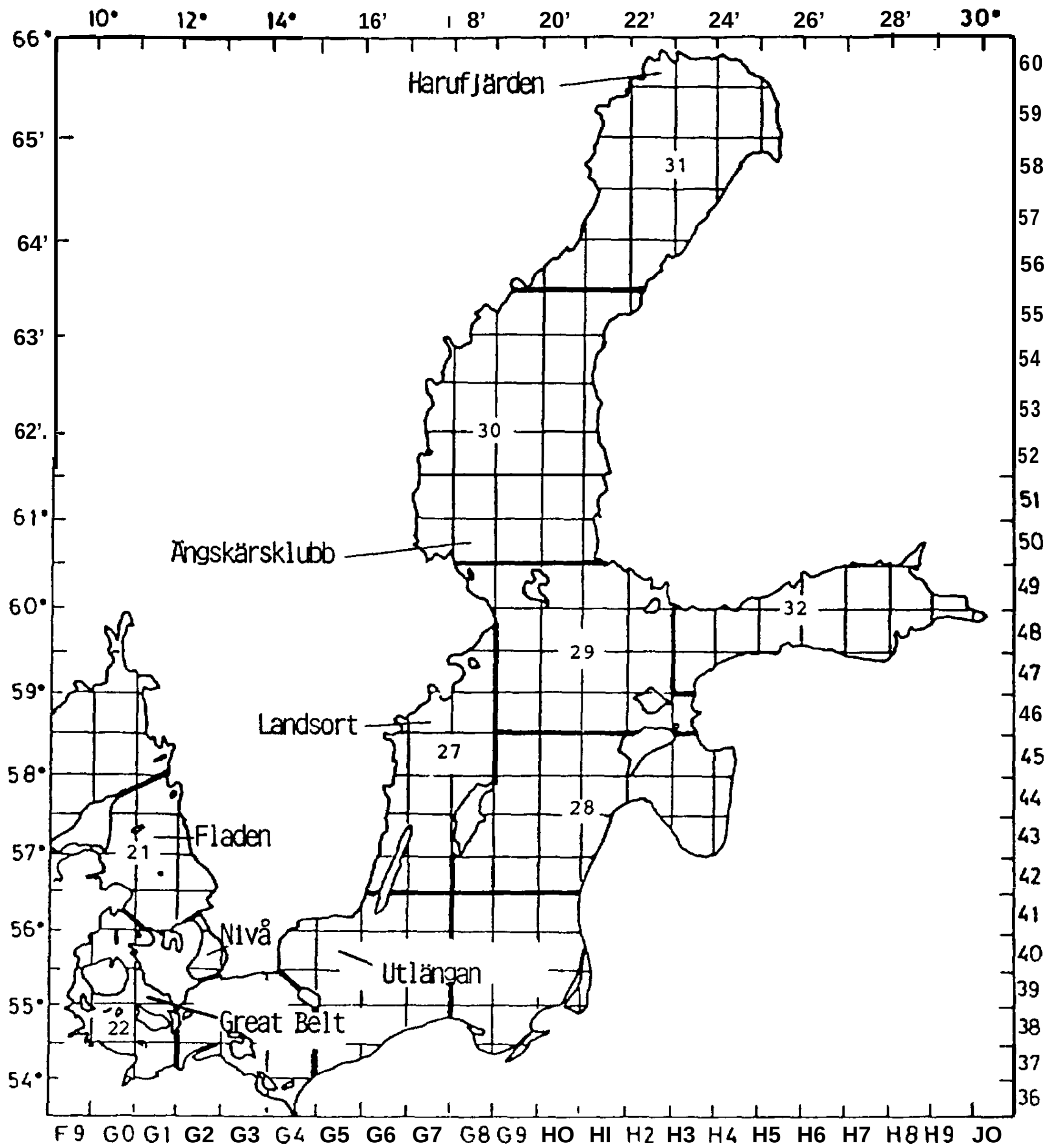

Figure 6. Sampling locations of the Swedish and Danish monitoring programmes.

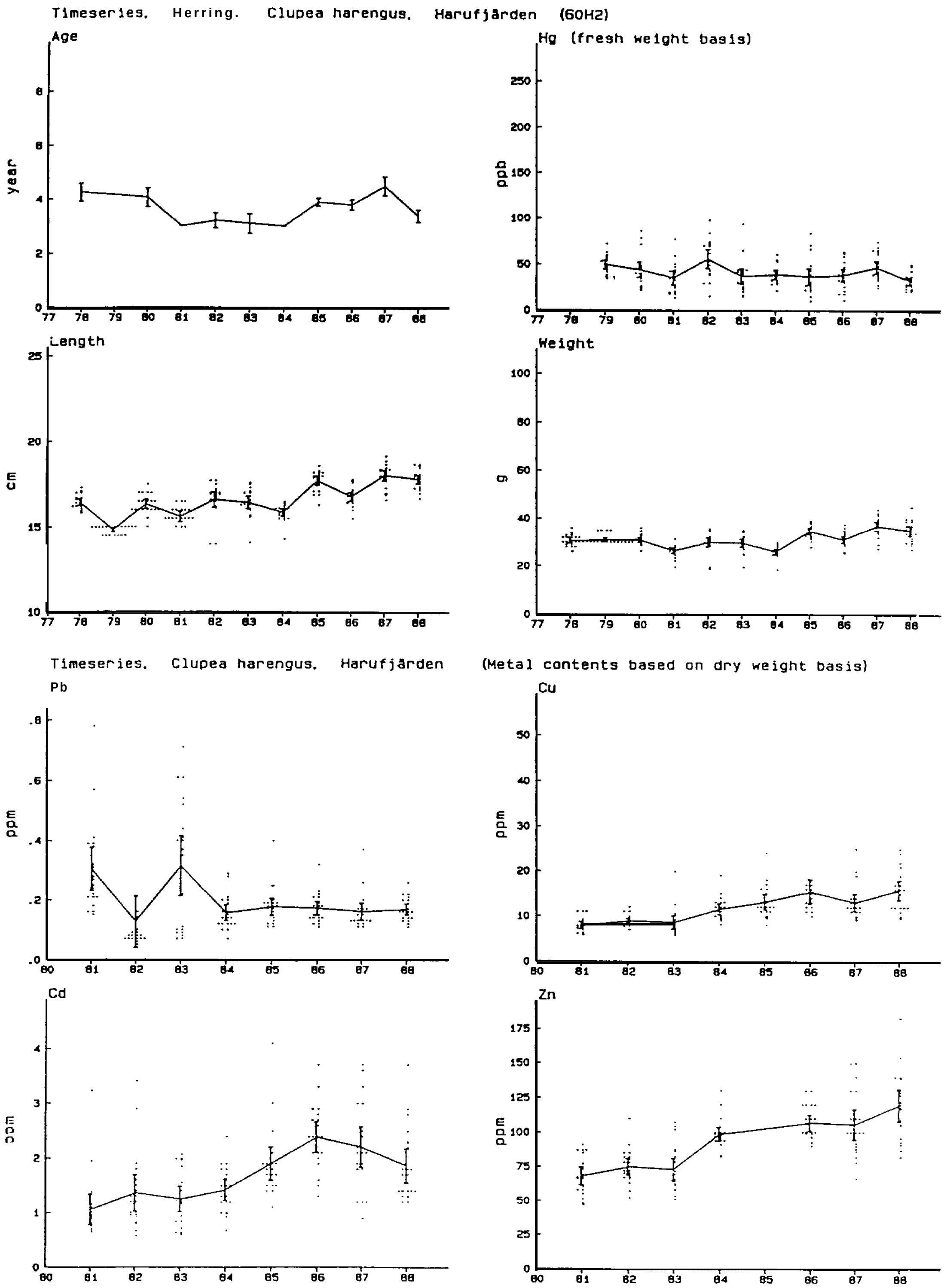

Figure 7. Trace element concentrations in herring from the Bothnian Bay, **Harufjärden** region (**60H2**). Hg concentrations (1979-1988) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1981-1988) on dry weight basis (liver tissue). **Age,** length and weight of fish (1978-1988) indicated.

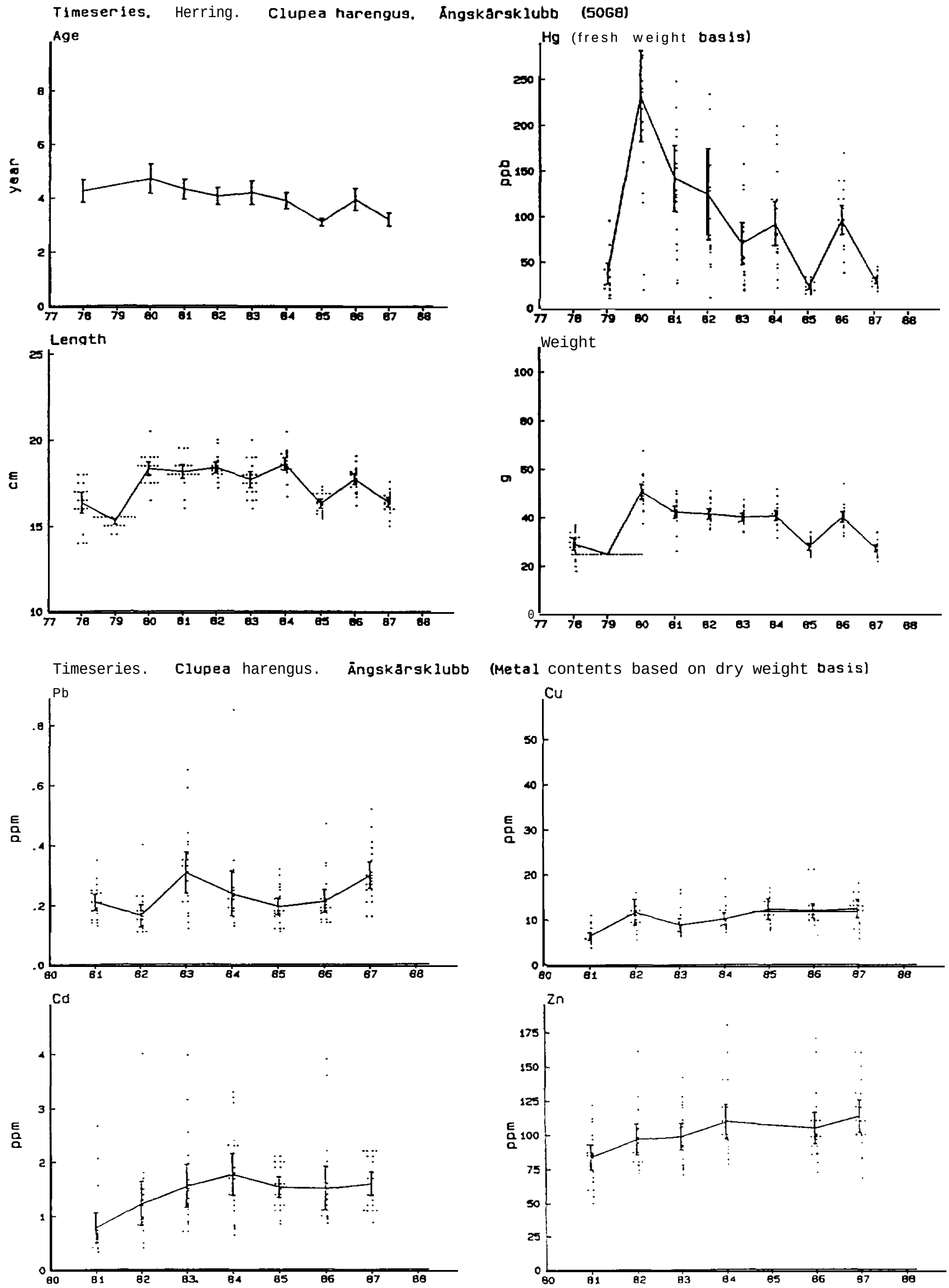

Figure 8. Trace element concentrations in herring from the Bothnian Sea, **Ängskärsklubb** region (50G8). Hg concentrations (1979-1987) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1981-1987) on dry weight basis (liver tissue). **Age,** length and weight of fish (1978-1987) indicated.

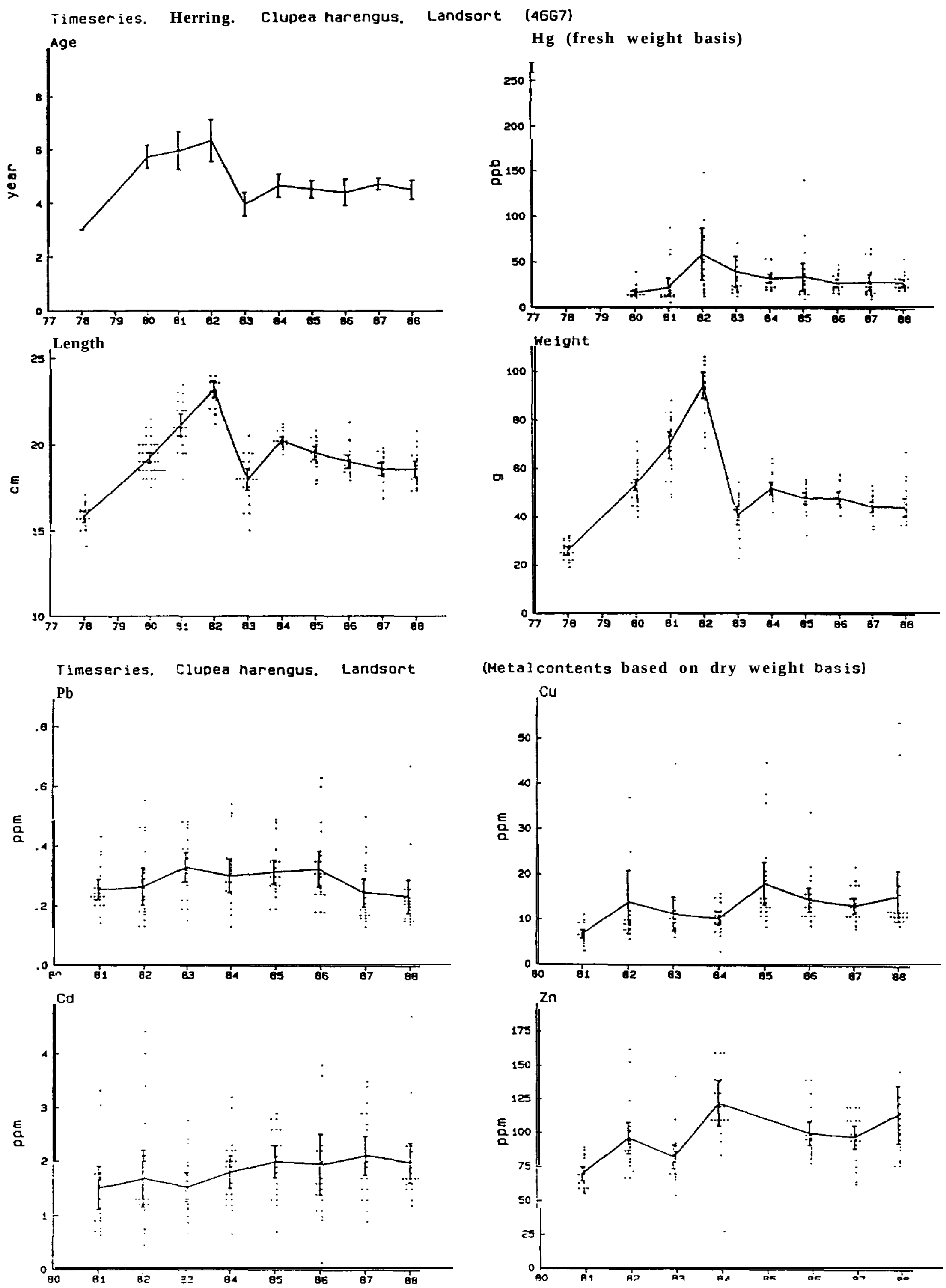

Figure 9. Trace element concentrations in herring from the Northern Baltic Proper, Landsort region (4667). Hg concentrations (1980-1988) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1981-1988) on dry weight basis (liver tissue). Age, length and weight of fish (1978-1988) indicated.

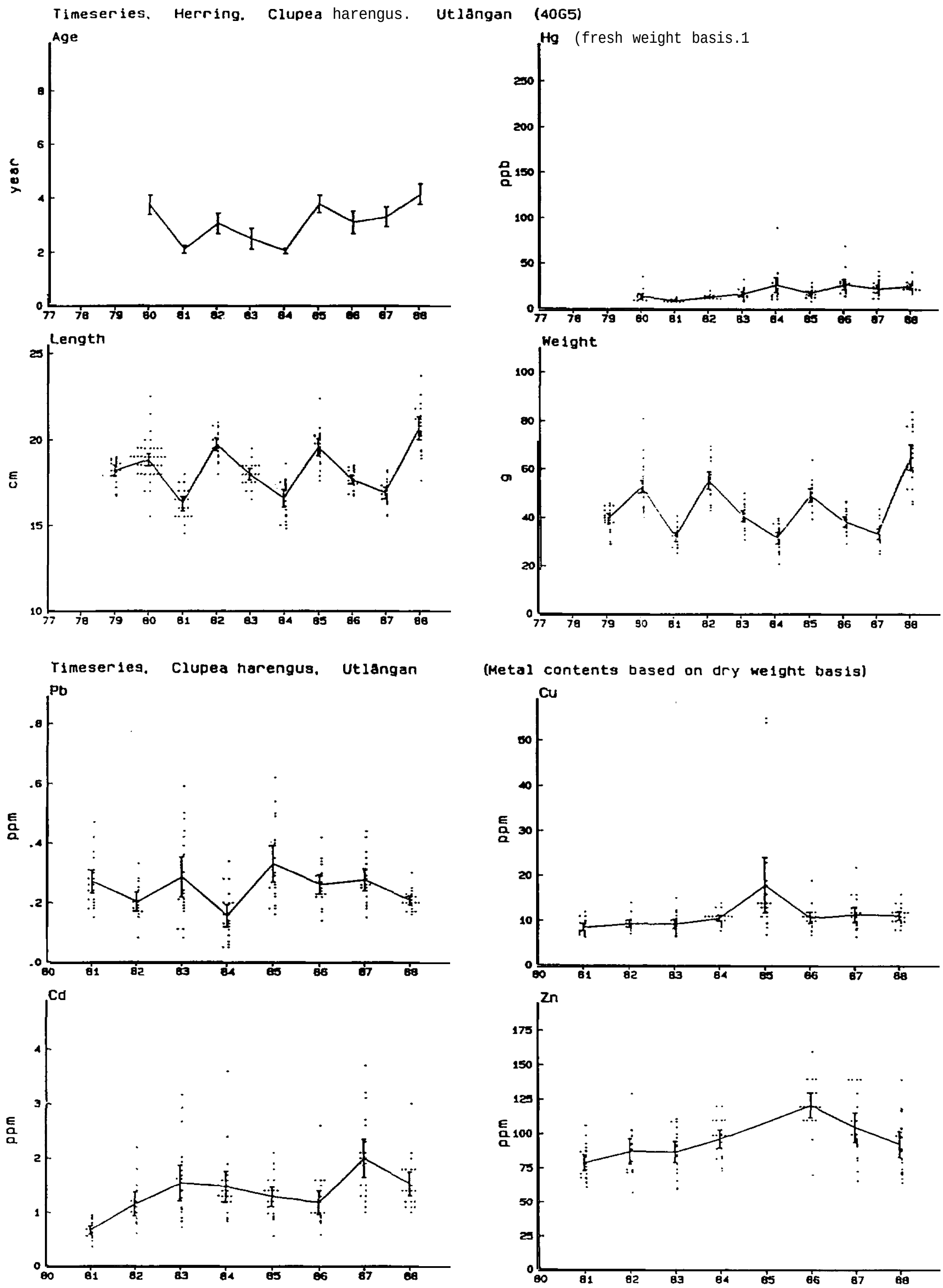

Figure 10. Trace element concentrations in herring from the Southern Baltic Proper, **Utlängen** region **(40G5).** Hg concentrations (1980-1988) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1981-1988) on dry weight basis (liver tissue). Age, length and weight of fish (1978-1988) indicated.

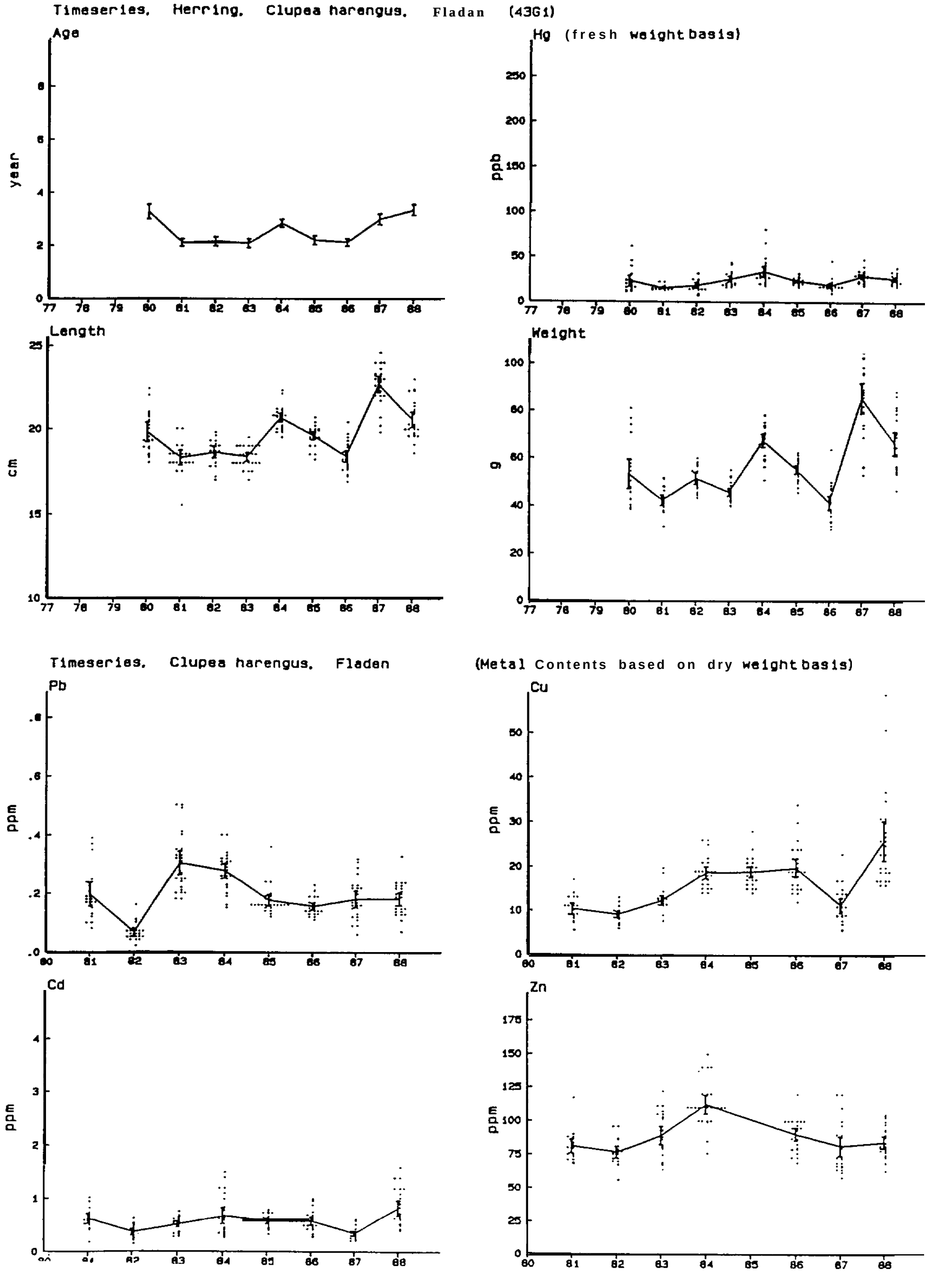

Figure 11. Trace element concentrations in herring from the Kattegat, Fladen region **(43G1).** Hg concentrations (1980-1988) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1981-1988) on dry weight basis (liver tissue). **Age,** length and weight of fish (1980-1988) indicated.

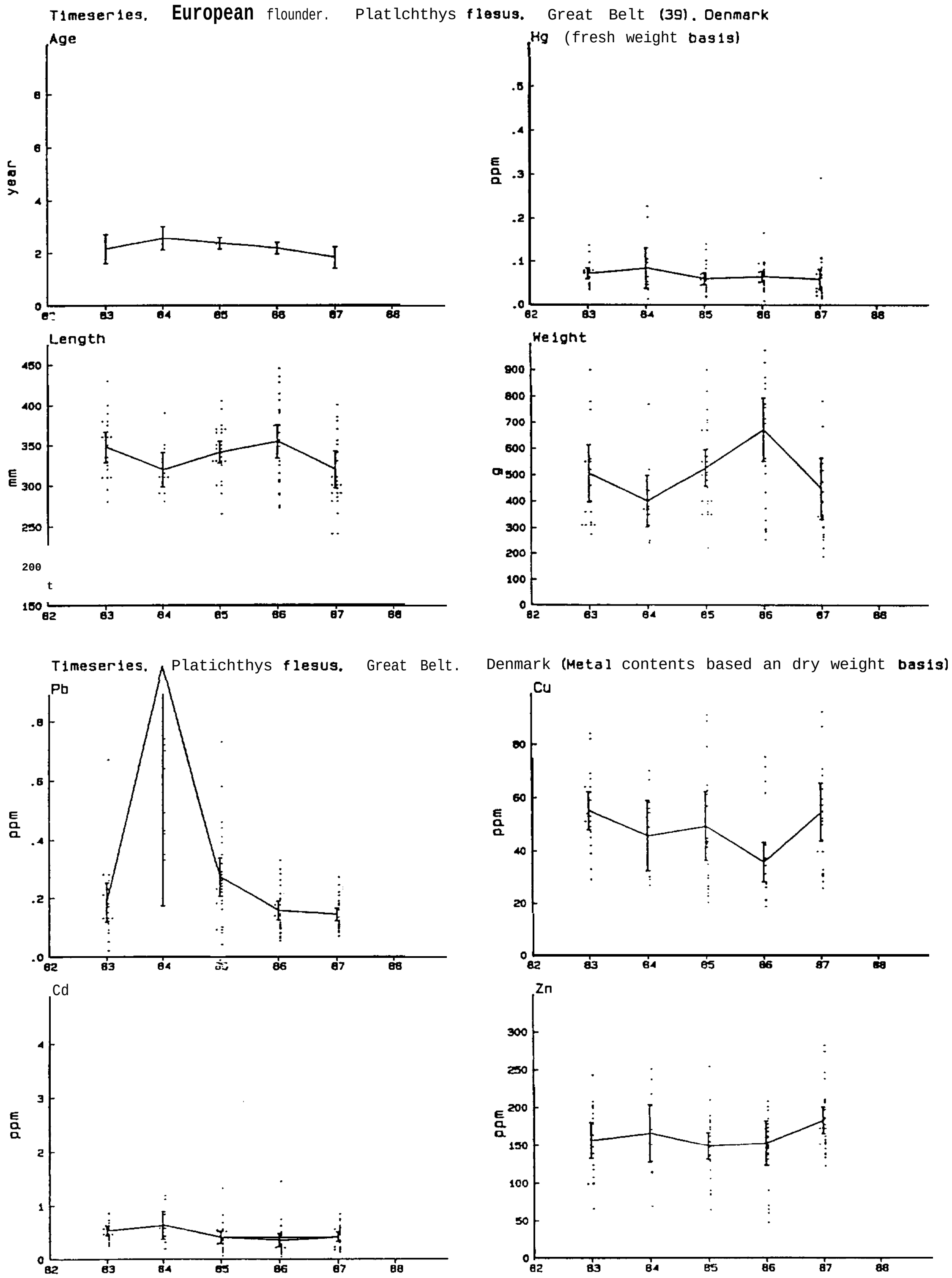

Figure 12. **Trace element concentrations in flounder from the Great Belt (39G1). Hg concentrations (1983-1987) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1983-1987) on dry weight basis (liver tissue). Age, length and weight of fish (1983-1987) indicated.**

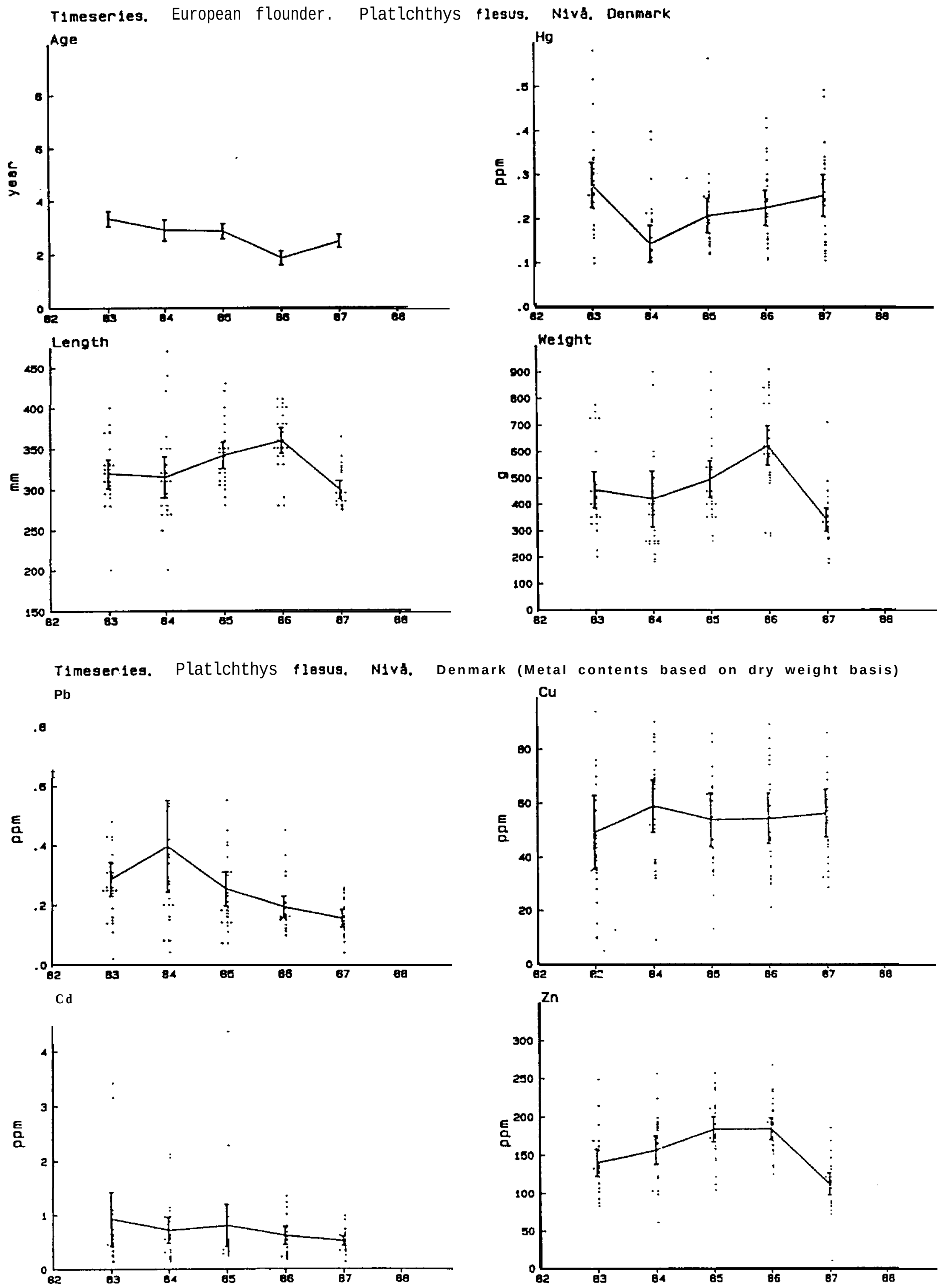

Figure 13. Trace element concentrations in flounder from the Sound, Nivå region (40G2). Hg concentrations (1983-1987) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1983-1987) on dry weight basis (liver tissue). Age, length and weight of fish (1983-1987) indicated.

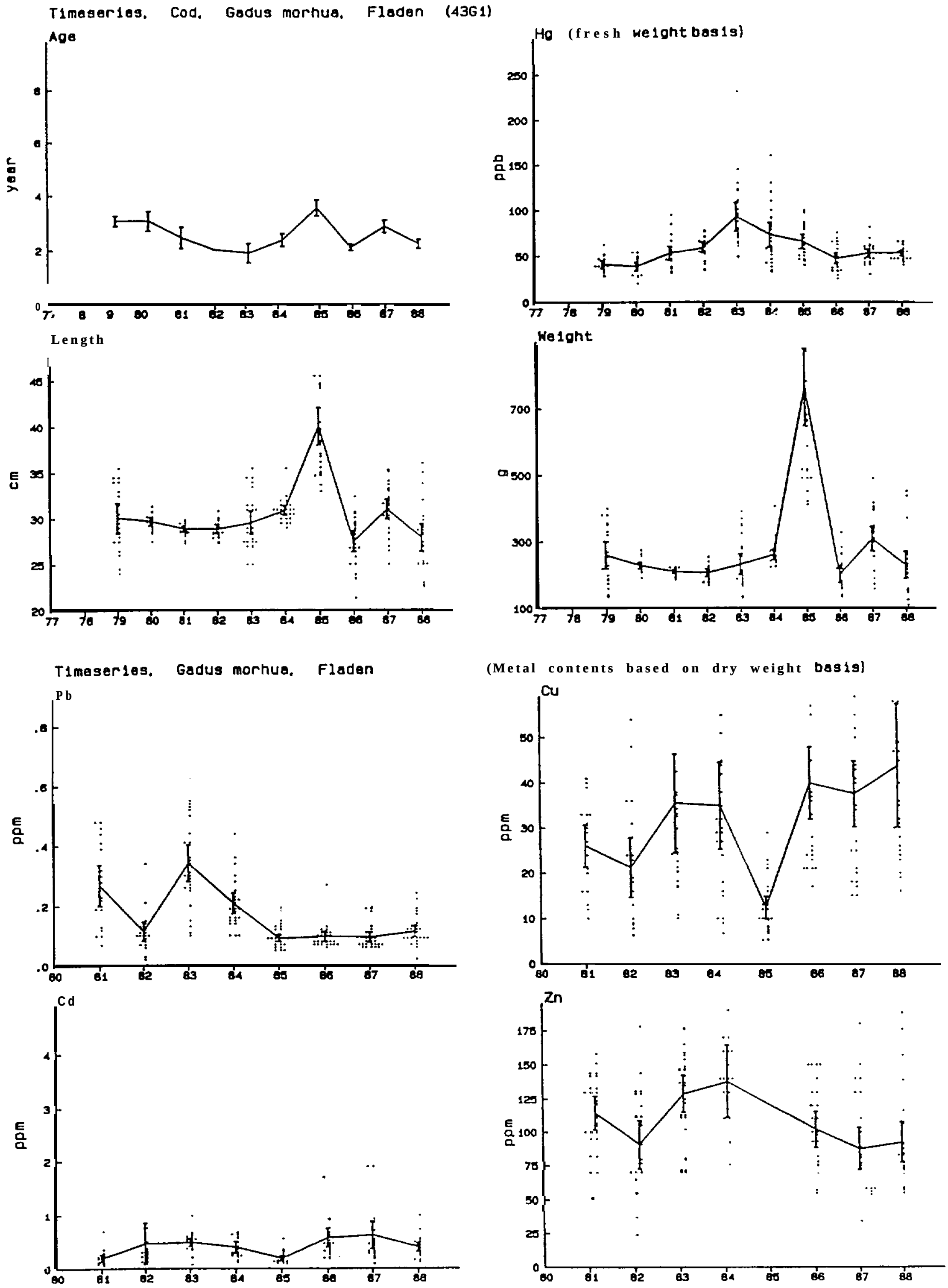

Figure 14. Trace element concentrations in cod from the Kattegat, Fladen region (4361). Hg concentrations (1979-1988) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1981-1988) on dry weight basis (liver tissue). Age, length and weight of fish (1979-1988) indicated.

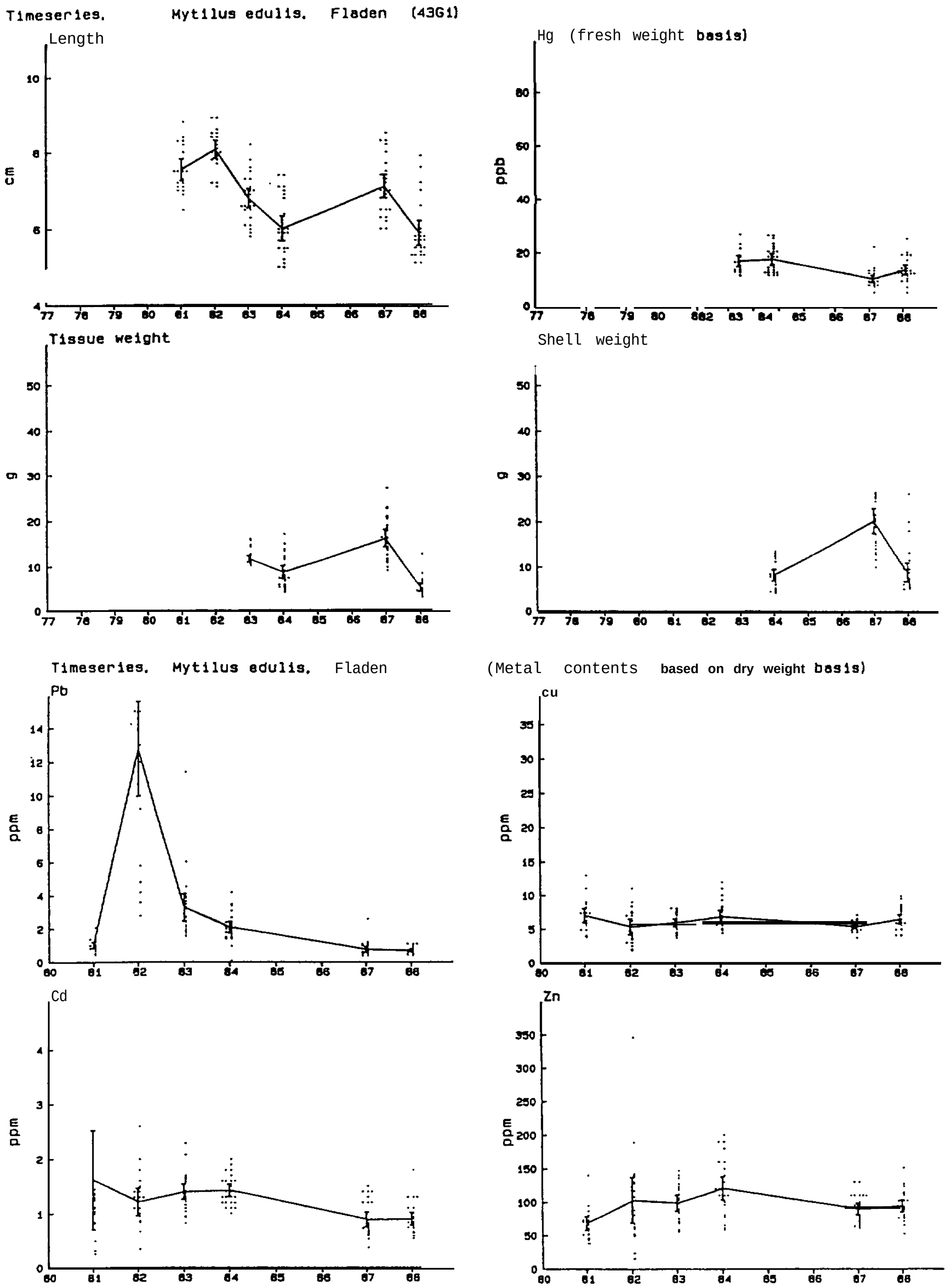

Figure 15. **Trace** element concentrations in blue mussel from the Kattegat, Fladen region **(43G1)**. **Hg** concentrations (1983-1988) on fresh weight basis (whole soft body); Cu, **Zn,** Cd and Pb concentrations (1981-1988) on dry weight basis **(whole** soft **body)**. **Length of** mussels **(1981-1988),** weight of whole soft body (1983-1988) and shell weight (1984-1988) indicated.

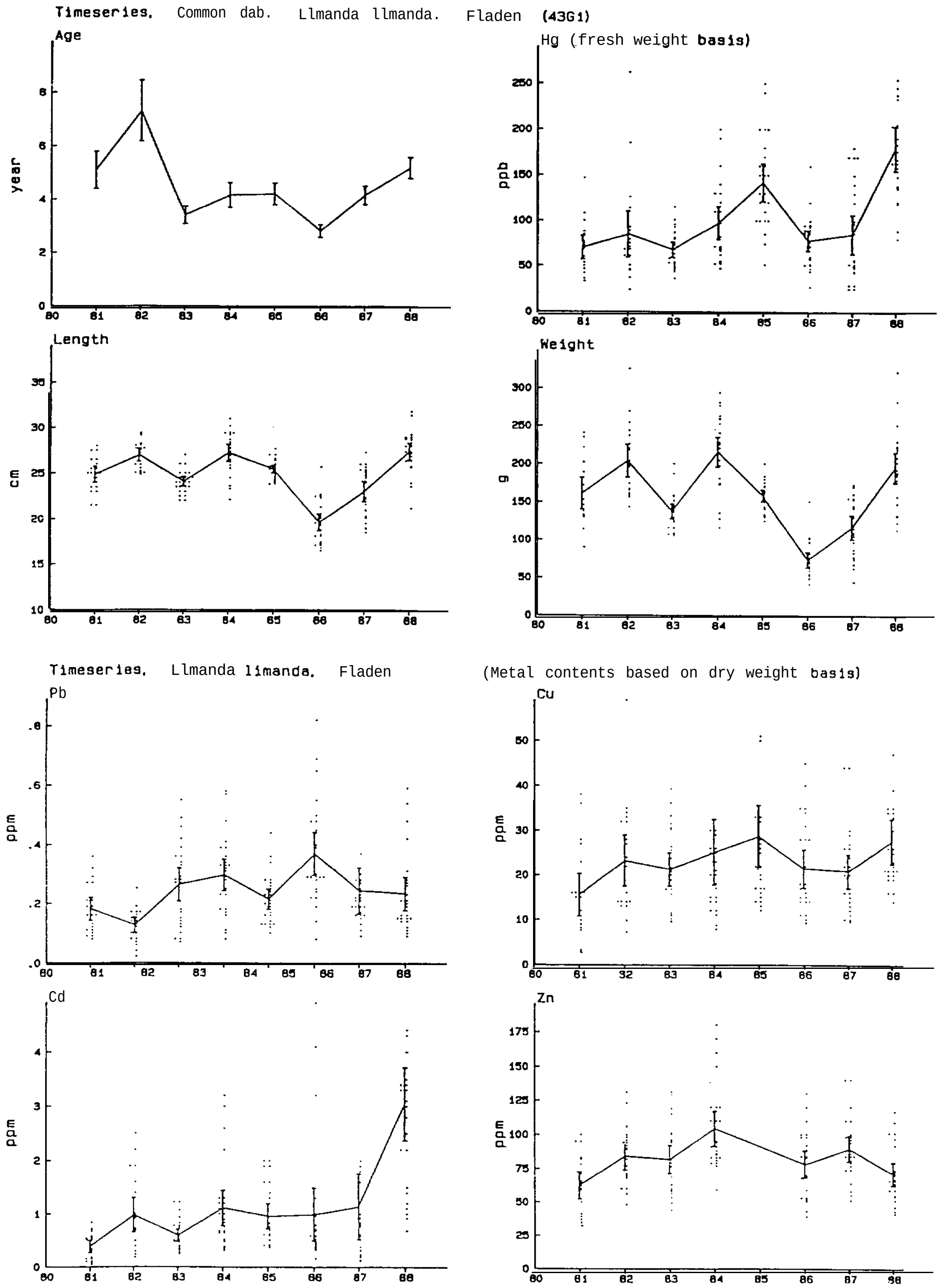

Figure 16. Trace element concentrations in common dab from the Kattegat, Fladen region (43G1). Hg concentrations (1981-1988) on fresh weight basis (muscle tissue); Cu, Zn, Cd and Pb concentrations (1981-1988) on dry weight basis (liver tissue). **Age,** length and weight of fish (1981-1988) indicated.

Figures 7 - 13 graphically show the annual mean levels, together with the 95% confidence intervals, of mercury, cadmium, lead, copper and zinc in herring and flounder respectively from different locations both of the Danish and Swedish marine environment. Plots (Figures 14 - 16) for cod, blue mussel and common dab *(Limanda limanda)* are also available for the Kattegat **(43G1).** Cd, Pb, Cu and Zn data are calculated on a dry weight basis of liver tissue analyzed, Hg data are based on wet weight analyses in muscle tissue. Information on some biological variables (length, weight, age) of the fish specimens analyzed is given in separate plots.

In several instances, there was evidence that at least part of the variation in mean metal concentration from year to year was due to a linear trend. However, there were often considerable fluctuations between years, which made **temporal trend** assessments more difficult. The pattern of contaminant variation between years must be interpreted with care. Principally, there could be a true long-term underlying trend, with large annual fluctuations around this, caused by variations in the environment under consideration. Alternatively, contaminant levels could have been rising for a certain period of time, followed by a decreasing period. Or levels could have changed from one long-term tendency to another.

In the following, some findings based on the aforementioned time series and on other investigations to be cited are summarized. It should be noted, however, that the data sets reported will be carefully statistically analyzed by the data originators in due course. Therefore, only tendencies which seem to be quite obvious (from pure inspection of the data) will be reported.

It is stressed again that due to uncertainties associated with analytical problems (see introduction) data on cadmium and lead concentrations in fish muscle tissue are not taken into account in the following section.

8.3.5 Lead

Similar trends in a number of different data sets were observed in the Kattegat (43G1, Fladen): cod, herring and blue mussel showed long-term changes with decreasing tendencies since the early 1980s. Data sets on flounder collected from the Sound (40G2) at station "Nivå" support these findings.

The concentrations of lead in flounder from the Great Belt (39G1) in 1984 showed a remarkable variance with some highly enhanced values. Unfortunately, only a limited number (11) of analyses had been reported from this year compared with 19-25 from the other years. Somewhat enhanced values from 1985 compared with 1986 and 1987 indicate that the high value reported in 1984 might be true. However, it is also possible that contamination during sample handling or analysis created systematic errors.

8.3.6 Copper and zinc

From the 1985 Baseline Study on Contaminants on Fish and Shellfish (ICES, 1988), from the aforementioned national contributions to the monitoring of the Baltic and from other relevant publications (Falandysz et al., 1984, Falandysz 1986 a,b,c, Tervo, 1987), it may be concluded that typically the concentrations of copper and zinc in muscle tissue of fish from the Baltic Sea are below 1 mg/kg and 10 mg/kg wet weight, respectively, whereas the corresponding liver data are a factor of about 5 to 10 higher. Generally, it can be stated that, compared with data from the North Sea, the copper and zinc levels reported are not significantly different in the Baltic Sea. Copper levels in Baltic herring have only recently increased considerably (by a factor of about 2.5 from 1987 to 1988) in the Kattegat area (43G1, Fladen).

A linear trend with slightly increasing tendency was observed both for copper and zinc in Baltic herring from the Bothnian Bay (60H2, Harufjlrden) since the early 1980s.

Zinc levels in Baltic herring had also increased in the Southern Baltic Proper (40G5, Utllngan) between 1981 and 1986, but significantly decreased thereafter.

8.3.7 Cadmium

Average Cd concentrations in flounder liver for the period 1983-1987 of the Danish trend monitoring programme exhibited significant differences between the Sound (0.72 mg/kg dry weight = approximately 0.16 mg/kg wet weight) and the Great Belt (0.44 mg/kg dry weight = approximately 0.1 mg/kg wet weight). The cadmium concentrations in flounder from the Sound decreased over the years considered, those from the Great Belt showed a downward trend from 1984 to 1985 and remained constant thereafter. As a consequence of the tendencies observed, both areas considered show nearly equal cadmium levels in flounder in 1987 (0.51 mg/kg dry weight = 0.12 mg/kg wet weight and 0.41 mg/kg dry weight = 0.09 mg/kg wet weight respectively). Such values were also found by Luckas et al. (1987) for flounder collected from the Kattegat, the Little Belt, the Kiel Bay and Bay of Mecklenburg.

Results gained from the analysis of several species (herring, cod, dab, *Mytilus)* collected from the Kattegat (43G1, Fladen) by Sweden were inconclusive with respect to long-term changes of contaminant accumulation pattern or site. Data on cod and herring showed cadmium levels fluctuating over the years, giving no evidence for a trend. Dab indicated an upward trend. Data sets on blue mussels had a decreasing tendency. Thus although the area has been studied rather extensively, a more general statement about the development of the contamination situation over time cannot be made.

An analysis performed on herring collected from the far north of the Bothnian Bay (60H2, Harufjärden) showed that the cadmium concentrations had increased substantially in recent years (with a maximum in 1986). A similar trend was observed for zinc and copper, as mentioned above.

The average concentration of cadmium (over the years 1983-1987) in the whole soft body of the blue mussel, ***Mytilus edulis***, from the Kattegat (43G1) was approximately an order of magnitude lower than in mussels from the Swedish coast in the Bothnian Sea (50G8).

A similar - although less distinct - gradient could be observed for Baltic herring investigated in the framework of the aforementioned Swedish programme. Concentrations of cadmium in liver tissue were on an average 0.53 mg/kg dry weight (average value for the period 1983-1987) in the Kattegat (area 43G1); they were, however, 1.59 mg/kg dry weight in the Bothnian Sea (area 50G8).

On 69 stations in the Belt Sea and Arkona Sea and the Southern Baltic Proper (ICES subdivisions 22, 24, 25 and 26), a total of 9633 cod ***(Gadus morhua)*** specimens was examined by Lang et al. (1987) for externally visible skeletal deformities. Investigations included the determination of cadmium concentrations in the liver and kidney tissue of 94 malformed and 317 normally developed individuals. Cadmium levels measured in the kidney and liver samples of cod from the whole area of investigation ranged from < 0.003 mg/kg up to a maximum of 0.178 mg/kg (kidney) and 0.276 mg/kg (liver) wet weight, respectively.

Irrespective of the health condition of the individuals analyzed, results revealed slight, but significant, higher contamination levels in the eastern and northern parts of the area investigated, thus indicating again a concentration gradient similar to the case of blue mussel and Baltic herring.

Information gained from the afore-mentioned investigations is that higher cadmium concentrations are found the further north in the Baltic samples were taken.

The dominant source of cadmium input into the Baltic Sea is from rivers (Baltic Marine Environment Protection Commission, 1987). Several studies have demonstrated an inverse relationship between dissolved cadmium concentrations and salinity (Huizenga et al., 1983, Balls, 1985, Kremling, 1987) in coastal mixing zones. This is mainly due to the fact that cadmium has a low particle reactivity (a weak tendency to be adsorbed onto suspended particulate matter in relation to other elements such as mercury or lead). Therefore, higher cadmium concentrations in biota collected from the northern sub-regions of the Baltic Sea may be associated with higher proportions of riverine water (i.e. higher cadmium input) in those areas.

A further consideration regarding the above-mentioned northwards increase in concentrations of cadmium in organisms is the consequence of salinity on the bioavailability of this element. The Baltic salinity decreases from about 20 PSU in the Kattegat to about 5 - 7 PSU in the Bothnian Bay. This change (gradient) in salinity causes changes in the chemical speciation of cadmium, which in turn may have an influence on the efficiency with which this element is accumulated by marine organisms. Phillips (1977), for example, found higher cadmium concentrations in blue mussels ***(Mytilus edulis)*** in areas where salinity was lower relative to other areas of higher salinity. It has not yet been proven that an inverse relationship exists between salinity and the uptake of cadmium by organisms at different trophic levels. However, it is evident from investigations carried out in the Baltic that the spatial decrease of salinity going northwards in the Baltic is accompanied by an increase of cadmium concentration in biota.

It is also debatable, whether the negative trend of salinity observed in nearly all sub-regions of the Baltic during the last decade (compare chapter "Hydrography") may be reflected by (inverse) upward temporal trends for cadmium concentrations in biota. Observations that cadmium concentrations in herring from the far north of the Bothnian Bay (6OH2, Harufjärden) have increased substantially during recent years may possibly be attributed at least to a certain extent to this phenomenon. However, it is recognized that this effect is probably superimposed by variations of the total metal load in that sub-region from discharges of metal producing/processing industries.

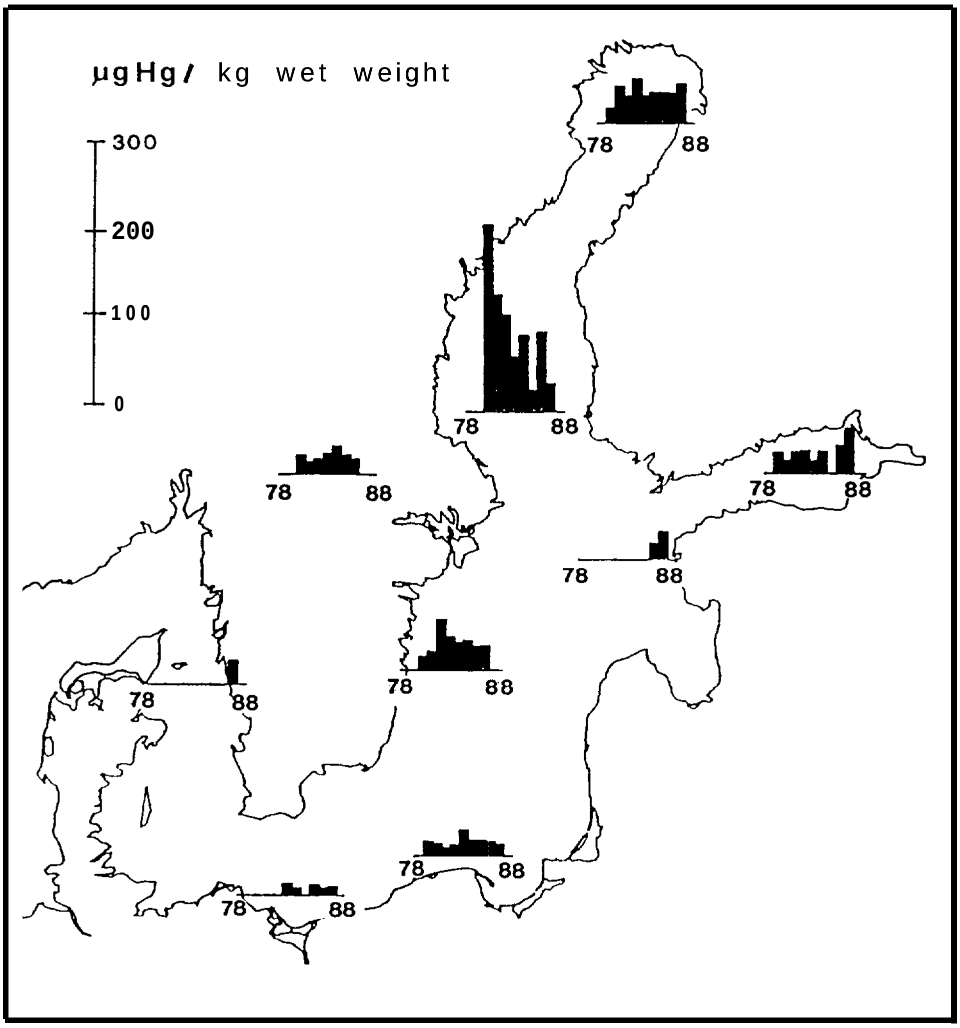

Figure 17. Graphical presentation of BMP data on levels of mercury in herring muscle for the period 1978-1988 received by the Finnish Institute of Marine Research.

8.3.8 Mercury

Relevant literature provides a considerable amount of information on mercury levels in marine fish and shellfish. In particular, data from the "Six-Year Review of ICES Coordinated Monitoring Programmes (ICES, 1984), and from review articles (GESAMP, 1986 and ICES, 1989) suggest that for most species the present background level is in the range of 0.05 to 0.15 mg/kg wet weight.

On the basis of these findings, monitoring data submitted by Baltic Sea States reflected in most instances concentrations approximate at background levels. This is also true for BMP data received by the Finnish Institute of Marine Research (compare Figure 17). Based on the information available there were only two areas of concern. At station Nivå in the Sound elevated *mercury* levels were measured in flounder with an average concentration of 0.22 mg/kg wet weight for the period 1983 - 1987. Findings suggest, together with what is stated above, that flounder from this area may still have been contaminated with mercury during recent years. In addition, Baltic herring from Ängskärsklubb (5OG8) in the Bothnian Sea also have higher mercury levels , although a considerable decrease of the concentration level could be identified.

However, this observation must be considered very cautiously, since it may - at least partly - be a result of sampling inconsistencies over the years. In other words, the structure of the data observed does not reflect adequately the temporal trend in that area. When herring is sampled for trend monitoring purposes, the age-dependent migration behaviour of this species must not be ignored. The young age classes of herring up to the time of sexual maturity are regarded as being stationary and thus representative of the area where they are collected. The older age classes however have shown to perform long-distance migrations. Accordingly, different age classes analysed bring different information. Therefore, it cannot be excluded that older specimens of herring collected from 1980 to 1984 at Ängskärsklubb may have visited the most severely polluted parts of Gävle Bay before they migrated into the area where they were caught. In contrast, the younger fish collected in the years thereafter may be assumed to have stayed mainly in the actual sampling area. Consequently, a critical evaluation of the time series leads to the conclusion that the decreasing tendency in mercury concentration may be less pronounced than suggested by these analyses, since it is probably superimposed by a "sampling effect".

In accordance with the monitoring programme of the Helsinki Commission, the concentrations of zinc, copper, lead, cadmium and mercury in Baltic herring, cod and two species of invertebrates (*Macoma baltica* and Saduria *entomon)* have been measured annually since 1979 in the Finnish sea areas (Gulf of Finland and Gulf of Bothnia).

During the five-year period 1982 - 1986, levels of the metals seem not to have changed considerably (Tervo, 1987). Data were also inconclusive with respect to existing differences in the spatial distribution of copper, zinc, cadmium and lead. Variations in the metal concentrations in biota within one sampling area masked possible differences between the sampling sites studied. However, in the case of mercury, at least for herring muscle tissue, significant geographical differences could be detected. In the Gulf of Bothnia, the average total mercury concentration (five-year average of the period 1982-1986) was 0.02 mg/kg wet weight, whereas the corresponding value in the eastern Gulf of Finland was 0.04 mg/kg wet weight.

As is the case with cadmium, the multispecies-approach (several species from the same area) applied to the Kattegat (4361, Fladen) led to inconclusive results. Herring and dab analyzed in that area indicated consistently slight upward trends. Cod, however, showed mercury levels fluctuating over the years, giving no evidence for a trend. The same observation was made for blue mussels.

8.3.9 Organotin compounds

Björklund (1989) provided results of a survey on the occurrence of organotin compounds in the Swedish aquatic environment. Concentrations of tributyltin ranged in samples of fish (flounder) from < 0.05 to 0.28 mg/kg dry weight, in samples of free-living blue mussels *(Mytilus edulis)* from 0.15 to 3.5 mg/kg dry weight, and in samples of other invertebrates from 0.05 to 4.3 mg/kg dry weight. Tributyltin occurred in blue mussels at essentially all locations investigated, which suggested that there was a comprehensive dispersal of these compounds in Swedish waters.Samples from the area of investigation at the west coast of Sweden, and locally also at the Baltic coast, showed the same kind of shell deformation as those observed within the tributyltin polluted areas of the southeast coast of England.

There is growing concern about the presence of organotin compounds in the marine environment. Namely, serious problems occur through the use of tributyltin (TBT) as antifouling agent in paints. In areas of restricted water circulation and intensive recreational boating, leaching of TBT from treated surfaces has caused adverse effects on marine life. It is recognized that water concentrations even at the ng/l concentration level of TBT can have lethal and sublethal effects on a wide variety of marine organisms, particularly on the sensitive early life stages of fish and shellfish (Laughlin et al., 1985, Thompson et al., 1985, Cardwell et al., 1986, and Waldock et al., 1987).

SUMMARY

In relation to previous assessment periods the trace element data basis is still rather limited and allows only very preliminary conclusions on trends.

Regarding particle-associated metallic trace elements it is still very difficult to compare data from different laboratories, owing to methodological uncertainties. There is an urgent need for intercomparison and standardization of the various procedural steps in obtaining data on particle-associated trace metal concentrations, particularly with respect to sampling methods and pre-treatment of the sample.

In view of the high variability in the Baltic Sea ecosystem, there is an inadequate pool of data which does not provide reliable information on the particle-associated concentrations of trace metals in different water masses and at different seasons, on the metal content of suspended particulate matter, and on the nature of the particulate matter in terms of origin and mineralogical and grain-size composition.

Gradients and differences observed in the concentrations of particle-associated trace metals, concentrations of the suspended particulate matter, and the distribution pattern in the metal contents of the different types of particles may reflect both anthropogenic influences and natural processes that cause redistribution and generation of suspended particulate matter of different composition and properties.

Trace element concentrations in fish and shellfish have not changed remarkably since the early eighties. Generally, it can be stated that mercury values in biota do not significantly differ now from those in the North Sea and the North-East Atlantic. Compared with present background levels, elevated mercury concentrations were only found at the station Nivå in the Sound, and at the station Ängskärsklubb in the southern Bothnian Sea. For the latter station, however, a considerable decrease of the concentrations could be identified during recent years.

Fish and shellfish from sampling locations in the Kattegat and the Belt Sea showed tendencies for decreasing lead concentrations. These findings are supported by tendencies of dissolved lead concentrations in Baltic Sea water. It is possible that this is already an effect of the increased use of unleaded petrol.

A tendency of increasing cadmium concentrations in biota when going in northerly direction in the Baltic Sea could be identified. The prime factor behind the fact that higher cadmium concentrations were found the further north samples were taken in the Baltic is obviously an inverse relationship between salinity and cadmium uptake. The Baltic salinity varies from about 5-7 PSU in the Bothnian Bay to more than 20 PSU in the Kattegat. These changes in salinity are accompanied by changes in total concentrations, in the chemical speciation and bioavailability of cadmium.

Observations that cadmium concentrations have increased in fish from the station Harufjärden in the northern part of the Bothnian Bay during recent years may possibly be attributed at least to a certain extent to the decreasing salinity in the Baltic between 1979 and 1988. However, this effect is obviously small in comparison with the influence of still existing discharges of industries.

Because of first adverse effects (shell deformations) found for blue mussels from the Swedish coast, there should be growing concern about the presence of organotin compounds (namely tri-butyltin) in the marine environment

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9. ORGANIC CONTAMINANTS

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ABSTRACT

This report is a part of the Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1984-1988; Background Document. Data collected on organic contaminants according to the Baltic Monitoring Programme together with results reported in open literature on monitoring and research on fate and effects of organic contaminants in the Baltic Sea have been compiled and evaluated.

9.1 INTRODUCTION

9.1.1 General

Contamination of the sea by stable organic contaminants originates from widely different sources. Atmospheric and riverborne transport adds to the point source discharges. Complex industrial effluents add to municipal discharges containing chemical products used in households. Storm water is contaminated from road traffic, etc. The worldwide transport and use of petroleum inevitably leads to discharges and accidental spills contaminating the environment. The problems caused by stable organic compounds have been a subject of attention since the discovery of DDT and later PCBs as widely distributed pollutants in the marine environment.

The knowledge gained from years of monitoring and research on these two compounds has resulted in restrictions on the use of them at least in the industrialized countries. This is mirrored by decreasing levels in the Baltic Sea.

Experience from these "stories" has resulted in the development, in several countries, of methods for testing and evaluation of chemicals and regulations for their release into the environment. Nevertheless the list of substances which now need to be monitored has increased and the analytical chemists will "successfully" find new environmental contaminants also in the future.

There is certainly a great need for stronger activities to control the production and use of stable organic chemicals in order to reduce their input into the environment. This can be done only by simultaneous **international** product control regulations and national efforts to control the emissions from all responsible sectors. To guide and support this work and to function as an additional defence barrier, regular chemical and biological monitoring, as well as an unprejudiced search for harmful known and "unknown" substances in the environment, is necessary.

In this chapter the state of contamination of the Baltic Sea by stable organic compounds is reported. Contaminants of petroleum hydrocarbon origin are discussed separately as are pesticidal components. The third sub-chapter deals with other types of organic contaminants including also unidentified halogenated compounds analyzed as adsorbable organic halogens (AOX) or extractable organic chlorine (EOCl).

The second stage of the Baltic Sea Monitoring Programme for the period 1984-1988 recommends the following hazardous substances to be analysed in selected organisms: DDT, DDD, DDE, PCBs and toxaphene. The preferred animal species are: herring, cod, blue mussel, flounder, *Saduria* and common shrimp. In addition to this, the Contracting Parties are recommended to include also at least one species of marine birds. Sea water and sediment samples should be analyzed for petroleum hydrocarbons (PHCs) and chlorinated hydrocarbons only on a tentative basis.

9.1.2 Harmful contaminants

There are certain properties which characterize environmental contaminants regarded to be generally distributed and potentially harmful. These substances or their metabolites are persistent, bioaccumulable and toxic.

Persistence implies a certain capability to withstand chemical, photochemical and biological degradation, rendering a substance a long life which makes possible its transport and distribution over wide areas. Persistence will also imply that contaminants will accumulate in the environment over the years.

Bioavailability implies that a substance is taken up in biota. Special concern is given to bioaccumulable substances which means that high concentrations can be built up in biota. Even worse are substances, which due to persistence, will also biomagnify in top predators. The lipophilicity of non-polar substances affects their distribution to the fat of the organisms. This is one important factor for their inclination to bioaccumulate. The steric structure of the molecule will affect its prerequisite to adsorb to detritus and thus it will influence both its transport in the environment and its bioavailability.

Bioaccumulating substances are per definition concentrated in biota. Thus the levels found in organisms are influenced by the density of the biomass in the area. A **release** of a certain amount of, for instance, PCB implies higher body burdens in fish in an oligotrophic area than in an eutrophic area.

The ultimate ecological effect caused by the release of a chemical substance depends on which ecological society and its organisms are exposed and under which climatological and hydrological circumstances the substance reaches the environment. The toxic effect might then be acute or chronic for the various species exposed.

The coastal areas are especially exposed because of the proximity to land-based pollution sources. Further, the major part of the organisms in the sea are to some extent coastal bound and harmful effects in these areas give rise to ecological effects also in the central Baltic Sea.

To summarise: a non-persistent, bioavailable and toxic substance ought to give an ecological effect only in the vicinity of the discharge, whereas a persistent substance will affect also remote areas of the sea.

9.1.3 Bioaccumulation

The bioaccumulation of a substance in biota depends on the output to the environment and the amount of biota into which it can be absorbed (Olsson and Jensen 1975, Olsson 1977). In addition the amount of particles to which the substance can be adsorbed is important. Compiling comparable **data** on DDT and PCB in fish from the Baltic, a recent report has stressed the counteracting effect of eutrophication on levels of bioaccumulating organochlorines in biota (Neuman et al. 1988). At a similar output of persistent bioaccumulating chemicals, the area having the highest density of the biomass ought to have the lowest amount per gram biomass. The compiled data indicate that this phenomena occurs in the Baltic. Data on DDT and PCB levels in plankton presented by Roots and Peikre (1981) might indicate a similar phenomenon and higher levels are generally found in off shore samples compared with samples collected closer to the coast. In future baseline and monitoring studies, the influence of the density of the biomass on recorded levels in organisms has to be considered.

For any substance found in biota where partition between water and lipids is a part of the uptake mechanism, the content of lipids in the organism is important. In ICES and HELCOM monitoring programmes, the fat content is determined when organochlorines are analyzed. The importance of this has been shown by Perttilä et al. (1982) in their study on Balti herring.

In organisms, the reactions leading to partition equilibrium, however, are time dependent. The clearance half-life for various organochlorines investigated in goldfish, for instance, varies between 10 to 60 days (Brcggemann et al. 1981). Thus a rapid decrease of the lipid concentration in fish during the spawning period might imply that the concentrations of organochlorines increase in the body lipids during this -Cod.

Investigations on the seasonal variation of PCB levels in roach during a seven-year period, however, have also shown that increased waterflow in the contributories of a Swedish lake imply an increase of the PCB levels in the body lipids of the fish (Olsson et al. 1978).

Field data show that the organochlorine levels are generally higher in the lipids in spring samples of Baltic herring compared to specimens collected during autumn (Olsson and Reutergårdh 1986). Similar results have been shown on perch collected in a Baltic Proper archipelago (Edgren et al. 1981). The discussion above indicates that the higher levels of PCB observed during spring can have more than one explanation.

The results show the importance of having comparable collection periods when comparing monitoring data. In the HELCOM monitoring programme, early autumn is chosen as the recommended collection period.

9.1.4 Analytical methods

The analytical techniques and instrumentation for measuring organic contaminants in environmental samples is under rapid development. The analytical chemist faces greater difficulties the lower the levels of contamination are in the respective matrices. Thus the concentration of nonpolar, lipophilic environmental pollutants might be troublesome to monitor in the water phase with extremely low concentrations, while the higher levels in biota due to bioaccumulation are readily measurable.

Intercalibration exercises have repeatedly shown that in order to obtain comparable results between different laboratories it is necessary to use well-defined and agreed procedures. This means that most data reported in the literature on organic contaminants in the Baltic Sea should not be used for direct comparison. Regional gradients and time trends are thus possible to evaluate only from data produced by single laboratories using the same technique and preferably also the same personnel for several years.

In the interim report, Intercalibration Exercise on Organochlorine Compounds in Baltic Waters (Briigmann et al. 1989), it is concluded that before organochlorine compounds are included as obligatory determinands in the Baltic Monitoring Programme, several technical and personnel requirements have to be complied with. This deals with the availability of common certified standards, certain instrumental specifications, demand on experienced personnel and common analytical procedures.

9.2 PETROLEUM HYDROCARBONS

G. Dahlmann[2]

Preliminary remarks

The chapter "Petroleum Hydrocarbons" of the Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea Area is a revised version of the corresponding chapter in the first assessment, kindly supported by its authors, E. Andrulewicz and K.-H. Rohde.

For the purpose of this assessment summary data about petroleum hydrocarbons (PHC), i.e. data which are achieved by means of spectroscopic methods and not followed by a more precise determination of the compounds detected, have not been used because these data cannot be used to assess the state of the sea.

Accordingly, the many gaps in the knowledge about input, fate, and effects of PHC in the Baltic Sea, mentioned in the first assessment, have seemingly increased. However, in addition to a more precise and scientific interpretation of the monitoring results achieved in the past, this could lead to a better understanding of the role of PHC (not Only in the Baltic Sea) and support the application of more sophisticated analytical techniques.

9.2.1 Introduction

Petroleum hydrocarbons (PHC) in environmental samples comprise a very complex mixture of thousands of organic compounds with different behaviour and thus different effects on marine life. There is a broad range of substances, from harmless n-alkanes, some of which are even produced by marine organisms, up to the very toxic and in part carcinogenic aromatic and heterocyclic compounds.

The term PHC comprises crude oils and its refined derivatives, which contain different amounts of different types of hydrocarbons, e.g. alkanes, cycloalkanes and aromatics, both alkylated and parent structures. Compounds containing oxygen, nitrogen, sulphur, and various metals (Ni, V, Fe) are also present.

However, PHC also include hydrocarbons formed, during the pyrolysis of any carbon-based fuel, e.g. polycyclic aromatic hydrocarbons (PAH) which are also present in petroleum. Aromatic hydrocarbons from combustion sources are characterized by a less degree of alkylation than aromatics from crude oil, but this degree is temperature dependent. In addition, aromatic hydrocarbons, even PAH, are generated in refining operations and enriched in products from, for example, cracking processes.

Once released into the environment, all of these compounds are subjected to continuous and variable changes due to bacterial degradation, photo-oxidation etc. Accordingly, there are basic methodological problems in monitoring PHC contamination; a single representative parameter for PHC contamination cannot exist. It is sometimes even difficult or not possible to differentiate between PHC of anthropogenic and biogenic origin.

Nevertheless, there is a need to determine PHC contamination because of the common worldwide use of petroleum. Efforts have been made for more than 20 years to determine this kind of pollution. The correct scientific way to analyse mixtures is to determine their single components separately. With regard to PHC, this is hitherto an *ever* growing challenge for analytical chemistry.

Petroleum hydrocarbons in the Baltic Sea have been the subject of several extensive new overviews e.g. by Jorgensen (1985), Granby (1987), Baltic Marine Environment Protection Commission – Helsinki Commission (1987). The present work takes advantage of these reviews.

9.2.2 Input of petroleum hydrocarbons into the Baltic Sea

Threatening picture

Petroleum hydrocarbons enter the marine environment as a result of marine transportation, municipal and industrial waste deposition, through runoff, atmospheric input, offshore production and from natural sources. A worldwide estimation of the input is given by the International Maritime Organization (IMO 1981, Table 1).

Table 1. Estimated inputs of petroleum hydrocarbons to the Baltic Sea and to the world oceans.

	IMO 1981		Jørgensen et al. 1985	
	world oceans		Baltic Sea	
	%	1 000*t/y	%	t/y
Transportation				
chronic	35.6	1 050	5.2	4 000
acute	14.2	420	2.6	2 000
Refineries			2.4	1 800
Municipal waste water			15.4	12 000
Industrial waste water	40.0	1 180	0.9	700
Urban runoff			13.1	10 000
Rivers			58.8	45 000
Atmosphere	10.2	300	1.3	10 000
total (t/y)		2 950 000		76 500

The Baltic Sea is an area which differs considerably from the average world ocean. In particular, the fact that the Baltic Sea is a land-locked basin with a water volume that is small compared to the oceans makes it more susceptible to damage. Natural seeps are lacking, and offshore oil and gas production is still in its infancy. Transportation of oil and intensive shipping activities may be a larger source of PHC than in other seas, but the highest input figures are from land and atmospheric fallout.

These considerations are reflected by the figures given by Jørgensen et al. (1985) in Table 1. Melvasalo et al. (1981) estimated the total annual input of PHC to the Baltic Sea at 50-100 thousand tonnes, which is in good correlation with the overall input, estimated by Jorgensen et al. (1985). Enckell (1986) made an estimation based on national reports submitted to the Helsinki Commission. She estimates the annual input of hydrocarbons to the Baltic Sea to fall within the range of 21,000 to 66,000 tonnes. If these estimates are correct, the Baltic Sea receives up to ten times as much PHC relative to its volume as does the average ocean.

General remarks about the input estimates

According to the definition of PHC, given in chapter 9.2, input estimates for PHC should not be over-interpreted as the chemical composition of PHC from the different sources is different, in principle. Even the physical state of single contributions is different. Therefore, input estimates of PHC do not only include errors due to methodological uncertainties, but touch a fundamental problem. Figures for the input of substances into the sea, in principle, can only be given separately for each single substance. The quantitative comparison of PHC from different sources might be misleading because the qualitative differences are particularly great.

The aim of an input estimate for PHC is the assessment of the seriousness and the scale of the input5 from the various sources. However, a survey like this compares, to a great extent, like with unlike. Moreover, since the noxiousness of the load from the various sources is fairly unknown and by no means proportional to the amounts estimated, it would be wrong to base far-reaching conclusions concerning the seriousness of the PHC contamination from the different sources from their estimated contributions. "The need for conclusions still calls for better analytical methods and a better knowledge of the impacts, which faces us with a lot of not only methodologic but also philosophic problems" (Enckell 1986).

9.2.3 Analysis of petroleum hydrocarbons

Background

The *major* methods used for measuring petroleum hydrocarbons in the marine environment are: infrared spectrometry (IR), ultraviolet fluorescence spectrometry (W-F), high pressure liquid chromatography (HPLC), gas chromatography (GC) and combined gas chromatography-mass spectrometry *(GC/MS)*. In this order, a higher specificity, i.e., ability to differentiate PHC from other hydrocarbons, is achieved up to the possibility of being able to determine each single compound separately by GC/MS. Accordingly, these methods are also ranked according to a higher expense and higher costs of the instruments. In addition, corresponding extraction and clean-up techniques are used to increase the specificity *of* these methods.

In Baltic laboratories, IR and W-F techniques were most commonly applied in the 1960s and 1970s. There was a tendency to switch from IR to W-F during the 1970s. Later studies also used the other techniques (GC, HPLC and GC/MS).

There have been two main intercalibration exercises on petroleum hydrocarbons in the Baltic countries (Anon. 1977, Baltic Marine Environment Protection Commission - Helsinki Commission 1982). The second intercalibration - limited to the W-F technique - yielded comparable results and, consequently, this technique was recommended as a screening method during the first and second stages of the Baltic Monitoring Programme (Baltic Marine Environment Protection Commission - Helsinki Commission 1980, 1984). Moreover, the monitoring of PHC by W-F was obligatory during the second stage of the BMP, and is now tentative again during the third monitoring period (Baltic Marine Environment Protection Commission - Helsinki Commission 1988).

General remarks about the W-F method

In order to avoid the obvious confusion about the usefulness of the UV-F method and about the inclusion of the parameter PHC into the BMP, the value of the W-F method has to be discussed in more detail.

The "Kiel workshop", 1981, recommended the use of UV-F as a screening method for determining the PHC pollution of seawater (Baltic Marine Environment Protection Commission - Helsinki Commission 1982). Screening of a sea area means looking for "hot spots", i.e., areas of higher burden of contaminants.

The advantages of the UV-F method for PHC screening are obvious. The UV-F method is simple, sensitive, inexpensive and fast, and thus provides an effective tool for surveying large sea areas. Its principle is mainly connected to aromaticity. When excited at distinct wavelengths, aromatic compounds emit W-light at longer wave lengths. Thus, with regard to the PHC contamination of seawater, the most harmful substances of petroleum are measured. In addition, because aromatic compounds are only rarely produced by marine organisms - in comparison to other biogenic hydrocarbons - the specificity of W-F for PHC is increased markedly over the IR technique.

There is no doubt that a summary method such as UV-F is not adequate for determining the very complex and changing composition of hydrocarbons in marine samples. Increased fluorescence values have only to be regarded as a warning signal, and more sophisticated techniques like HPLC, GC and GC/MS must be used to find out the reason for the increased fluorescence intensities, i.e. the sources of the fluorescing compounds.

Regarding the very different compositions of the sources of PHC and the changes due to variable environmental factors, the composition of a marine sample differs drastically from the composition of the crude oil used as standard, except in the very rare cases of direct inputs where an appropriate standard can be chosen. Therefore, W-F should be used only for screening in a monitoring strategy, followed by more detailed investigations of the composition of selected samples by more sophisticated methods. In this sense, W-F saves time and capacities for the more detailed investigations.

PHC monitoring by W-F should be continued in the BMP, in order simply to achieve a starting point for a joint PHC monitoring. There is no time to hesitate, until a perfect solution of the problem of PHC determination is found.

Rearing in mind that the W-F screening greatly reduces the number of samples which has to be analysed in more detail, it should be possible to find solutions if the appropriate techniques are not available. Research projects within the framework of international co-operation are urgently needed.

9.2.4 Monitoring results

Petroleum hydrocarbons in water

There is a multitude of IR and W-F data from all Baltic countries over the past years. Owing to the fact that these data have not been followed by a more precise description of the composition of the hydrocarbons found, they have to be critically looked upon in an evaluation (see above). Table 2 is given only in order to show the historical technical development. An overview over extensive investigations in the years 1980 to 1987 is given in **Figure 1** (from Poutanen 1988).

Table 2. Summary values of PHC, achieved by spectroscopic methods.

Measurement period	Method	Concentration $\mu g/l$	Source of information
1969	not given	300-1000	compiled by Tervo
1970-1975	IR-spectr.	50-100	(1980) from
1971-1972	"	50-100	various sources
1973-1974	"	100-200	
1974	UV-F-spectr.	1	
1976	"	1-3	
1977-1979	"	0.3-4.1	
1978-1979	"	0.2-9.4	
1980-1982	"	0.2-14	Rohde & Briigmann (1985)
1981-1982	"	0.4-21	Law & Andrulewicz (1982)
1980-1983	"	O-7-18.5	DHI (1981-1989)
1983-1988	"	0.7-3.3	
1981-1985	"	1.0-1.6	Carlberg (1987)
1980-1987	"	0.1-2.5	Poutanen (1988)
1984-1987	"	1.0-2.3	Talvari (1988)

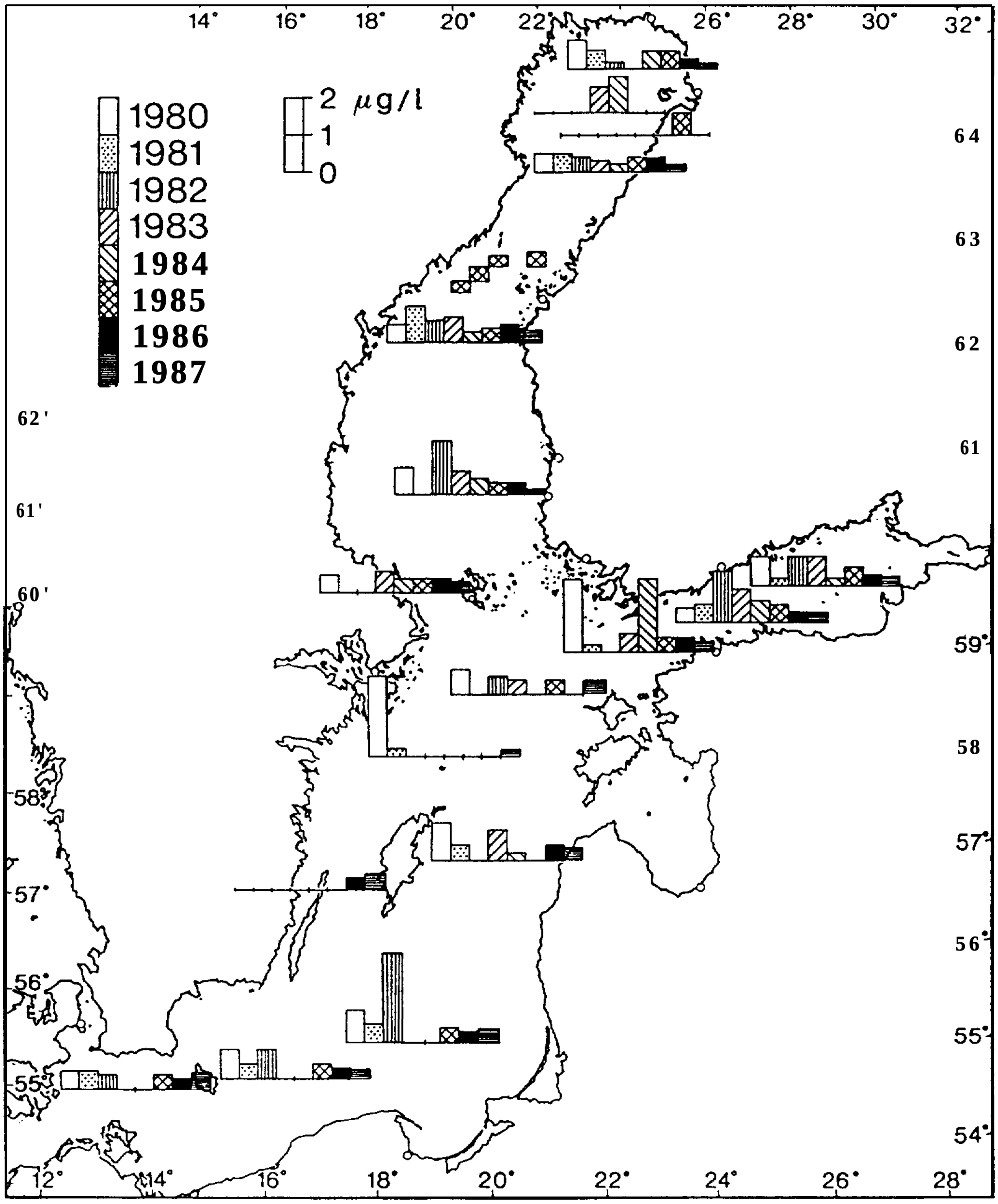

Figure 1. Total hydrocarbon concentrations (W-F values) in the time period 1980-1987 (from Poutanen, 1988).

Theobald (1988) found that up to 90% of the measured fluorescence in samples of Baltic seawater (n-hexane extracts) originates from polar, i.e. oxygen or nitrogen containing compounds, even though a part of these compounds may have been derived from non-polar, even petroleum, precursors, e.g. by photo-oxidation or bacterial degradation. However, in addition, a further inspection of the "real" hydrocarbon part of the samples shows that - unlike in petroleum - unsubstituted polycyclic aromatic compounds like phenanthrene, fluoranthene, pyrene etc. (PAH), predominate (DHI, 1989). These compounds originate mainly from combustion processes and enter the marine environment by atmospheric fallout or land or river run-off. Since the ratio of PAH to alkanes in these mixtures differs drastically from that of the crude oil used as standard oil, there is overall a large uncertainty in UV-F values from the Baltic implicating that isolated values are generally of limited value.

Nevertheless, some preliminary conclusions, drawn in the First Periodic Assessment from W-F results (Baltic Marine Environment Protection Commission - Helsinki Commission 1986), seem now to be verified, when samples are analyzed in detail. The even distribution of these values in all areas and depths, i.e. similar values are found in areas of high shipping traffic in the western Baltic as well as in open waters of the Baltic Proper, indicate the high contribution of the atmospheric input (DHI 1983). The suggestion that lower UV-F values in summer are the result of higher evaporation and higher bacterial degradation (DHI 1984) correlates with the latest results (Theobald, pers. comm.) showing a higher proportion of non-polar components in winter compared to summer.

Broman et al. (1988) found that the concentration of combustion products adsorbed on suspended matter in the Stockholm archipelago is higher in the winter-spring period than during the summer due to increased emissions and more extensive washout of land-deposited PAH during the melting of snow. Higher W-F values indicating a higher PHC pollution in coastal areas, e.g. in the Bay of Gdansk (Law and Andrulewicz 1982), the Gulf of Finland (Tervo 1980), the coastal area of GDR (Rohde and Briigmann 1983) and the Kattegat area (Jorgensen et al. 1985), have to be verified by more detailed investigations. Nevertheless, corresponding investigations in sediments (Law and Andrulewicz 1982, Granby 1987), bivalve molluscs and seston (Broman et al. 1985, 1988) confirm the rapid decrease of PHC contamination from coastal, especially urbanized areas, towards open waters.

Table 3. Comparison of the mean concentration ranges of PHC (UV-F values) in seawater from different sea areas, before and after correction by subtraction of the polar (non-hydrocarbon) fraction.

Sea area	concentration range µg/l	polar fraction %	corrected conc. range µg/l
Elbe estuary	4-40	48	2-20
North Sea	0.4-3	67	0.12-1
Baltic Sea	1-4	90	0.1-0.4
Atlantic Ocean	0.1-0.3	52	0.05-0.15

These examples show, that the PHC screening by UV-F is worthwhile, but that conclusions drawn from isolated W-F values, neglecting their relative character, must be wrong. The comparison of the mean ranges of W-F values from different sea areas (Table 3 , from Thebbald 1988), shows a somewhat higher range for the Baltic Sea than for the North Sea, but indicates about a tenfold burden compared to the Atlantic Ocean. When corrected by a subtraction of the polar part, the mean concentration range for the Baltic is only about half of the mean range of the North Sea and about twice of the mean range of the Atlantic Ocean. The relationships have changed drastically due to the high contribution of polar material to the fluorescence intensity in Baltic seawater. However, even these relationships might be far from reality because of the different compositions of the hydrocarbon mixtures in different sea areas. Combustion products, however, dominate in nearly all samples from all sea areas (DHI 1988).

Even less can be said about the variations of the PHC concentrations of Baltic seawater with time, despite the presence of extensive collections of W-F data from regular monitoring since the early 1980s. The figures given by Carlberg (1986), Poutanen (1988), Talvari et al. (1986), DHI (1983 - 1989) all indicate only small variations over the years, without any tendency - unfortunately, such a stagnancy over the last decade can never be proved.

Latest results of seawater monitoring are given by the Deutsches Hydrographisches Institut (DHI 1989). In 1987, the Baltic Proper was investigated, in addition to the routine monitoring in the Belt Sea, which is conducted once a year. The UV-F screening yielded very evenly distributed values in the western Baltic as well as in the open waters of the Baltic Proper (Figure 2). Therefore, samples from a few stations (marked by arrows in Figure 2) were chosen, representing the inner bays of Eckernförde and Liibeck as well as the open sea, and analysed in more detail by means of GC/MS.

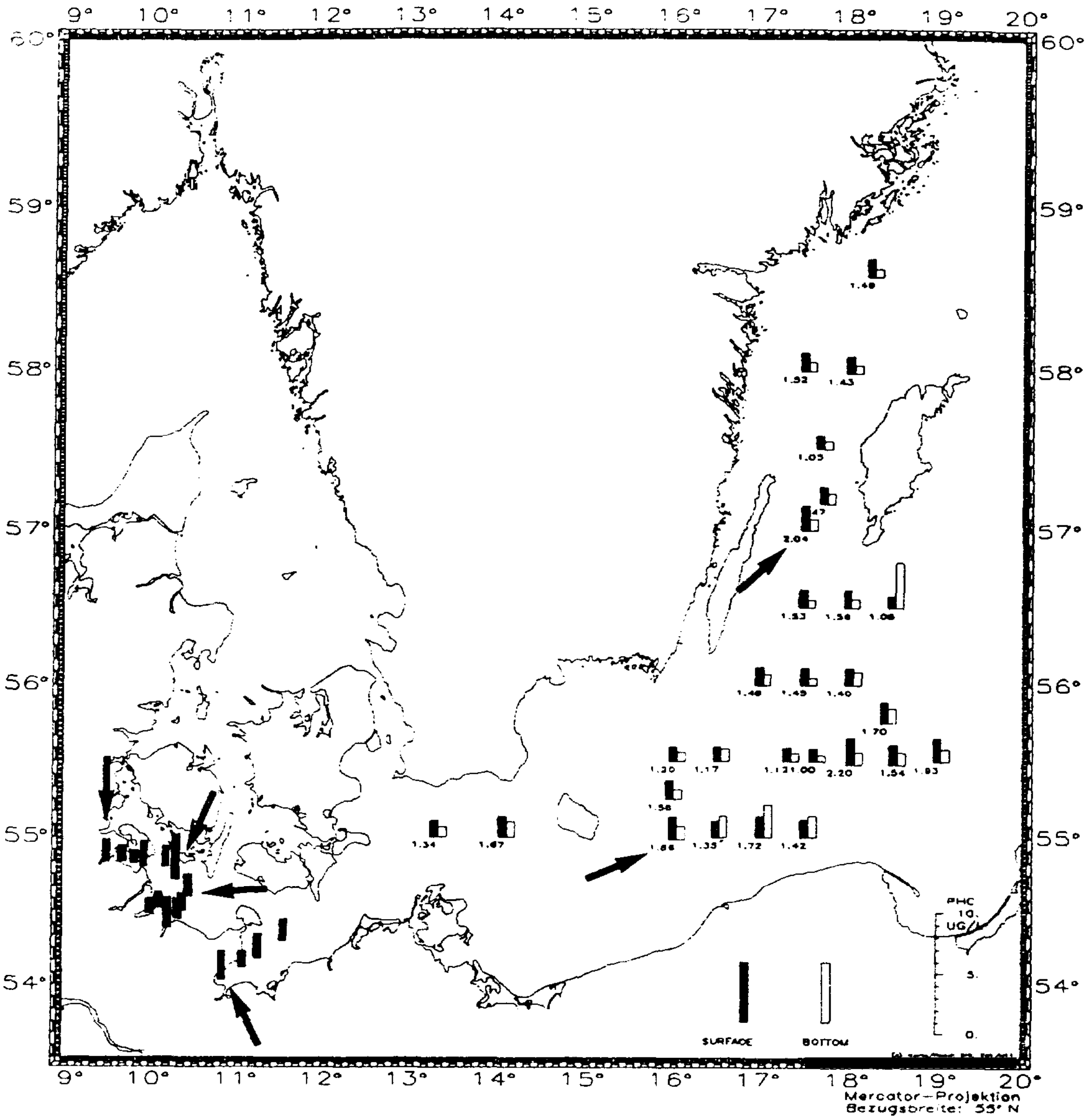

Figure 2. Total hydrocarbon concentrations (UV-F values) in August 1987. Stations from which samples are analyzed in more detail by GC/MS are marked by arrows, corresponding to the stations named OS... in Table 4 strictly from left to right.

Table 4. Concentrations of petroleum hydrocarbons (ng/l) in the Baltic Sea (OS), the German Bight (NS) and the Elbe mouth (near surface values, August/September 1987). For location of Baltic Sea stations, see Fig. 2.

station No.	OS 701	OS 706	OS 716	OS 715	OS 11	OS 19	Elbe	NS12	NS 21	NS 23
n-C17	16,18	77,77	30,77	49	91,49	173	40,06	32,34	90,21	241,11
pristane	7,31	3,11	3,69	3,84	2,69	2,88	4,91	12,53	11,07	4,52
n-C18	15,34	6,19	8,88	9,75	5,62	8,39	6,95	3,15	15,51	15,29
phytane	6,32	3,79	1,75	2,01	2,58	1,69	8,47	4,11	13,09	3,42
naphthalene	1,78	1,77	2,39	2,21	1,14	1,24	13,21	7,33	10,1	26,86
2-M-naphth.	1,28	1,26	1,41	1.29	0,76	0,92	4,85	2,97	3,27	6,88
1-M-naphth.	1,01	0,83	1,17	1,04	0,51	0,65	2,79	1,79	2,09	4,82
acenaphthylene	0,08	0,11	0,11	0,21	0,06	0,07	0,71	0,11	<0,05	0,19
acenaphthene	0,25	0,38	0,38	0,51	0,21	0,28	1,22	0,26	<0,05	0,46
fluorene	0,46	l,l.l	1,12	1,03	0,67	0,55	2,44	0,82	1,04	1,65
dibenzothiophene	0,19	0,38	0,31	0,28	0,28	0,22	1,59	1,01	1,31	0,61
phenenthrene	1,41	1,96	1,77	1,68	0,93	0,89	7,91	1,03	1,32	1,37
6-M-phenanthr.	0,39	0,28	0,28	0,28	0,19	0,16	1,46	0,17	0,71	0,27
4-M-phenanthr.	0,61	0,39	0,35	0,38	0,27	0,21	1,83	0,25	0,93	0,27
2-M-phenanthr.	0,69	0,33	0.33	0,35	0,26	0,21	1,09	0,23	0,56	0,27
1-M-phenanthr.	0,51	0,41	0,41	0,37	0,47	0,41	1,19	0,35	0,79	0,27
anthracene	0,21	0,08	0,14	0,12	0,05	0,09	1,84	<0,05	<0,05	<0,05
fluoranthene	3,17	1,17	0,81	1,08	1,53	1,03	17,17	0,68	1,41	0,58
pyrene	1,68	0,65	0,36	0,51	0,43	0,29	16,37	0,42	0,96	0,27
benzo[a]anthracene	0,11	<0,05	<0,05	<0,05	0,11	0,08	4,84	0,05	0,12	<0,05
chrysene/triphenylene	0,24	0,12	0,09	0,09	0,16	0,12	9,29	0,11	0,09	0,06
benzo[b]fluoranthene	0,26	0,09	0,12	0.08	0,16	0,11	13,59	0,16	0,15	0.07
benzo[k]fluoranthene	<0,05	<0,05	<0,05	<0,05	<0,05	<0,05	<0,05	<0,05	<0,05	<0,05
benzo[e]pyrene	0,11	<0,05	<0,05	<0,05	0,05	<0,05	6,62	<0,05	0,06	<0,05
benzo[a]pyrene	0,05	<0,05	<0,05	<0,05	0,07	<0,05	6,96	<0,05	0,16	<0,05
perylene	<0,05	<0,05	<0,05	<0,05	<0,05	<0,05	3,28	<0,05	<0,05	<0,05
indeno[1.2.3-c.d]pyrene	0,06	<0,05	<0,05	<0,05	<0,05	<0,05	4,32	<0,05	<0,05	<0,05
dibenzo[a.h]anthracene	<0,05	<0,05	<0,05	<0,05	<0,05	<0,05	1,68	<0,05	<0,05	<0,05
benzo[g.h.i]perylene	0,07	<0,05	<0,05	<0,05	<0,05	<0,05	5,59	<0,05	<0,05	<0,05

The results are given in Table 4, which also includes values from some stations in the river Elbe mouth and the German Bight, for comparison. The predominant concentrations of n-C17, which increase towards open waters, clearly indicate its biogenic origin. It is one of the major constituents of natural hydrocarbons and mainly produced by marine algae (Clark and Blumer 1967). Contamination is indicated by the presence of a variety of aromatic compounds, which concentrations range from about 3 ng/l to less than 0.05 ng/l, with a slight decrease to more open waters. Unsubstituted aromatics, like naphthalene, phenanthrene, fluoranthene, pyrene etc. predominate, which originate mainly from combustion processes. The values from the Baltic Sea are in most cases similar to those found in the German Bight, with the major exception of the Elbe mouth station, which shows about the tenfold concentrations. In addition, higher concentrations of alkylated aromatics and higher boiling compounds in the river Elbe indicate an additional load of compounds of petrogenic origin.

Petroleum hydrocarbons in particulate matter and sediments

Owing to their relatively low aqueous solubility and hydrophobic character, most PHC entering the aquatic environment either are, or readily become, adsorbed to particulate matter in the water column. Thus, the transport and sedimentation of suspended matter is an important mechanism for the distribution and incorporation of PHC into the aquatic ecosystems.

Due to the fact that the content of biogenic organic compounds from detritus or living organisms is often very high in particulate matter and sediments, chemists involved in environmental monitoring are aware that the determination of PHC in these matrices requires special extraction, clean up, separation, and identification techniques. A simple screening technique, such as UV-F, is not appropriate.

Broman et al. (1988) studied the spatial and temporal distribution of 18 PAH compounds in the Stockholm archipelago by means of seston samples, which had been collected by sediment traps. PAH concentrations and fluxes exhibited a steep logarithmic decline with distance from urban areas. These results confirm the heavy impact on the bottom deposits in urban areas. PAH concentrations and fluxes were higher during the winter-spring period than during the summer due to increased emissions and more extensive washout during the melting of *snow.* Granby (1987) measured saturated hydrocarbons (p-n-hydrocarbons) in sediment samples from 16 stations around Denmark. The highest contaminant load (1342 mg/kg dry weight) was found at a station near Copenhagen. Higher contents in samples from stations with no direct pollution sources could be explained by higher sedimentation rates and a higher content of organic material, including biogenic saturated hydrocarbons.

Mattson and Lehtinen (1985, 1987) found "increased levels of petroleum hydrocarbons in the surface sediments of Swedish coastal waters". 61 of the sediment stations investigated by Rudling (1976) in 197411979 had been re-visited in 1982. The levels of p-n-hydrocarbons showed a statistically significant increase from 199 to 252 $\mu g/g$ dry weight. The main increase could be found in coastal areas where the main pollution sources are situated. The authors calculated a yearly increase of p-n-hydrocarbons in surface sediments of 8700 t in the area investigated. Thus, a first indication of an increase of PHC pollution in the Baltic Sea is given, which could be the result of the permanent deposition of PHC, mainly from urban run-off on the *one* hand, and low degradation rates due to low water temperatures and large areas of oxygen deficiency on the other hand.

Poutanen (1988) investigated sediments in more open areas of the Baltic. No clearly oil contaminated bottom sediments could be found. The composition of the background content of PAH (0.5 -7 $\mu g/g$ dry weight) indicated the atmospheric input as the main source in these areas.

Petroleum hydrocarbons in bivalve molluscs

Marine organisms as monitoring objects have the advantage over seawater and sediments that at the same time one obtains an assessment of the bioavailability of contaminants. In addition, in some cases it will be possible to interpret the effects of the contamination by assessing the health conditions of the organisms. The mussel, as a stationary organism, is a suitable choice to measure local contamination loads. It accumulates organic contaminants relative to the concentration in seawater. By filtering large quantities of seawater, its exposure to soluble and particle associated toxic substances is high.

Granby (1987) used bivalve mussels as biological indicators of the PHC contamination along the Danish coastline of the Baltic. The highest levels of saturated hydrocarbons (p-n-hydrocarbons) and 15 selected PAH were found in mussels from Charlottenlund near Copenhagen, in the Kattegat off the Limfjord and at Rønne, Bornholm (108 to 39 mgPHC/kg wet weight; 36 to 111 μgPAH/kg wet weight). The high contents at these locations could be connected to local sources, such as large effluents from waste water or high shipping traffic. In addition, mussels from these stations showed a lower lipid content and a poorer health condition than mussels from open waters. Samples collected far away from local sources showed background levels of PAH, which mainly originated from atmospheric input.

These findings are in good correlation with the results achieved by Broman and Ganning (1985). By investigating mussels from 23 different locations in the Stockholm archipelago the authors could find a gradient in PHC contamination from the inner archipelago zone, near the urbanized area, to open waters (saturated fraction: 87 - 5.6 μg/g wet weight; aromatic fraction: 28.2 - 2,9 μg/g wet weight). Diffuse, continuous discharges from multiple sources, e.g. municipal wastewater, lake and river outflow, run-off from urban areas, industries, harbours etc., were assumed to be the cause of this gradient. By using a distinct bioconcentration factor, the authors assessed the concentration of PHC in the water. The values fall within the ranges of 20 to 3 μg/l (p-n-fraction) and 9 to 1 μg/l (aromatic fraction).

9.2.5 Oil spills

Exchange of information on spills of oil and other harmful substances in the Baltic Sea Area is the task of the Combatting Committee (CC) of the Helsinki Commission. It should be possible to prepare a complete list of all tanker accidents of the last decade, including their locations, types and amounts of spilled oil.

Regular oil spill surveillance in the Baltic is just at the beginning. A first attempt has been made to fulfil1 part of the CC obligation, namely to develop and establish airborne surveillance with adequate sensor systems for the detection of violations of the discharge provisions. Accordingly, the evaluation of surveillance statistics is not yet completed. A draft summary of national oil spillage reports from 1988 will be presented at the CC meeting in autumn 1990.

9.3 PESTICIDES

H. Gaul[2]

Pesticides are used for the protection of agricultural and forestry production in order to combat pests. Insecticides, fungicides, and herbicides serve this purpose. These deliberately designed toxic substances reach the **Baltic** Sea via rivers and from the atmosphere.

Even with orderly use of these substances on land the input into the sea can cause disturbances in the food web there, and can affect the use of the sea as a source of food for man.

DDT provides an example of these effects, the use of which led to a decline in numbers in fish-eating sea birds, and resulted in a ban on the marketing of cod livers from Baltic Sea catches in several countries some years ago.

The alarming effects of DDT were caused by the combined effects of four factors:

1) the toxicity of the substance
2) high persistence
3) high biological enrichment
4) the considerable amounts used.

All pesticides are deliberately designed to be toxic. The other characteristics are no longer general for the entire class of substances. However, to different extent they apply to the chlorinated hydrocarbons, for example: DDT, Lindane (-HCH), Dieldrin, Chlordane, Heptachlor, Mirex, and the substance mixture of chlorinated camphenes (Toxaphene).

As a measure of the stability (persistence) of a substance, the time is investigated up to which half of the substance has decayed under normal laboratory conditions (this means in the presence of water, oxygen, light, **moderate** temperatures and microbiological decay). These half-lives ($t_{1/2}$) are about 7 years for DDT, about 2 years for Lindane, for Chlordane and Dieldrin less than one year; for other pesticides they are even lower (Table 5).

The extent of the enrichment of a substance in biological material, calculated from the concentrations in the surrounding water and the concentrations in the tissues of the organism, is given in the BCF value (Biological Concentration Factor). As this value is dependent upon various influences, it can have a considerable range of variability, which extends from 250,000 to 4,000,000 for DDT, and from 200 to 500 for Lindane. Table 5 contains enrichment factors as they accommodate between the surrounding water and commercially exploited types of fishes.

Pesticides with low half-life values can be enriched by organisms in low levels only, because decomposition counteracts it. Substances with low BCF values generally cannot be so highly enriched that they would cause disturbances in organisms.

Table 5. **Half lives ($t_{1/2}$) and bioconcentration factors (BCF) of pesticides.**

DDT	7 a	250-4000 x 10^3	1
HCB	3 - 4 a	3 - 30 x 10^3	1
γ-HCH	2 a	200 - 500 x 10^3	1
Dieldrin	< 1 a	15 - 60 x 10^3	2
Chlordane	< 1 a	10 - 40 x 10^3	1
Parathion-Ethyl	5 d	5 - 200	3
Malathion	1 -2 d		4
Disulfoton	15 d		3
Fention	25 d		3
Methoxuron	20-30 d		5
Linuron	1 d	50	5
Atrazin	< 100 d	1	6
Simazin	50 - 70 d	1	6
24-D	200 d	100	7
2,4,5-T		1	7
Trifluraline	l h	40 - 4000	8

Source: Ecotoxicological Studies of the EEC Directive 76/464 List I Substances

1: Portmann (79) (DDT, HCB, HCH's, Heptachlor, Chlordane)
2: Butijn (77) (Aldrin, Dieldrin, Endrin)
3: Claus (85) (Parathion, Fention, Fenitrothion)
4: ISPRA (83) (Malathion)
5: Getti (85) (Linuron, Methoxuron)
6: ESF (86) (Atrazin, Simazin)
7: Micha (86) (2,4-D; 2,4,5-T)
8: EXCLOSER (86) (Trifluraline)

UNEP (OCA)/MED WG

Table 6. **The contamination of herring with organochlorine compounds (µg/kg fat).**

Area	HCB	a-HCH	γ-HCH	OCS	DDT	DDE	DDD	PCB-52	PCB-138	PCB-180
A	13.5	21.2	21.1	2.1	<0.5	39.9	<0.3	17.7	14.9	7.9
B	152	103	123	<4	0.7	1015	545	125	489	113

A : Shetlands Date: 06.88 Sample size: n=20 Fat: 7.12 %

B : Bornholm Date: 10.86 Sample size: n=20 Fat: 2.95 %

Reference: Krüger & Kruse 1988

Chlorinated hydrocarbon insecticides and fungicides have a diminishing significance in Europe; for the dangerous formulations there are restrictions on use and, for some, total bans. In non-industrialized tropical countries DDT is still in use because of its unsurpassed cost-effective benefits against carriers of tropical diseases like malaria. Nevertheless, the world production of DDT has decreased from 90 kt in 1963 to about 30 kt in 1979 and in the EEC countries amounts today to 7 **kt/a** only, 90 % for export.

Pesticides from the group of phosphoric acid esters and thiophosphoric acid esters are also produced in smaller quantities than previously. Substances like Parathion, Parathionmethyl, Malathion, Disulfoton, Fention, for example belong to this group.

The advantage of lower persistence with half-life values of a few days, and the lesser accumulation associated with it, is outweighed by the higher toxicity for mammals (and the human utilizer).

Insecticides on the basis of natural substances such as Nicotine or Pyrethrum components are less persistent and, in this respect, do not represent a risk to the environment.

Other insecticides and herbicides based upon carbamates and urea derivatives, such as Methoxuron or Linuron, have half-lives of a few days only.

Herbicides are gaining increasing significance in agriculture. If they are soluble in water in order to be taken up by the root systems, a high bioaccumulation is not possible. The lipophilic Trifluraline is an exception; this could be enriched if the short half-life of only one hour does not hinder it.

We have reason to assume that the short half-lives of the pesticides mainly employed at the present time prevent a risk from extending beyond river mouths into the open Baltic Sea (Niemirycz et al 1988, Cyberska et al. 1988, Zelechowska et al'1988).

We can proceed upon the assumption that the only organic contaminants that represent a threat to the entire Baltic Sea are those which have a high bioaccumulation potential and, as prerequisite thereto, possess considerable persistence. This applies only to certain chlorinated hydrocarbons, to which the following statements are confined for that reason.

9.3.1 DDT

Water

Intercalibration exercises on analyses of DDT and other organochlorine compounds in water have been unsuccessful in the past. There are no agreed methods for this type of investigation. Since 1975, reports have been given about analysis of DDT in sea water applying various methods. A comprehensive list of results up to 1981 were presented at the 13th Conference of the Baltic Oceanographers (Mohnke et al. 1982). These data were based upon gas chromatography using packed columns, an insufficient method for trace analysis in water.

The results are sometimes contradictory and any conclusions with regard to spatial or temporal trends can be deduced to a limited extent only (Brügmann et al. 1989).

Investigations using gas chromatography with capillary columns in 1983 revealed quantifiable concentrations of DDT in water of the western Baltic Proper and the Belt Sea (Fig. 3). These results were confirmed by a corresponding increase of the DDT residues in biota. Since then, the concentration in water has again fallen below the quantification limit (Gaul 1987, Gaul 1990).

Biota

Intercalibration exercises on analyses of DDT and its metabolites in biota were more successful than those in water because the concentrations to be measured are much higher due to bioaccumulation. If levels of residues in the same species, age class, and sex are compared in different regions of the Baltic, it should be possible to identify "hot spots" of pollution. Herring is available in all parts of the Baltic Sea and was therefore chosen to be an obligatory species to be sampled in Baltic Monitoring Programme (Fig. 4).

Though the residues in herring from the Baltic Proper are higher than those in the Kattegat or the Bothnian Bay, these differences between the average mean values are small if the variability of the data in single specimens in one year and age class is taken into account (Fig. 5).

The levels have not changed much since 1980, with the exception of the mentioned increase between 1983 to 1985 in the western Baltic and the Belt Sea (Fig. 5). This was due to the widespread application of DDT in forestry in one of the Baltic states. Since then, the concentrations have returned to their previous levels (Odsjd & Olsson 1989, Haahti & Perttilä 1988, ICES 1988b).

From the very even distribution of the DDT residues in herring, we can conclude that the main source of input of DDT into the Baltic Sea is the atmosphere. This suspicion is confirmed by the lack of indications of any relevant local inputs after most Baltic states had banned the use of DDT. The DDT concentrations in herring decreased by 90 % within 10 years. This is well documented for the southern Baltic Proper by Olsson et al. (1984). This decrease is confirmed, e.g., by a corresponding decline in human breast milk in Estonia (Roots 1986) and Sweden (Norén 1988).

Mammals

The final links in the marine food web in the Baltic Sea are - apart from fish-eating sea birds - the seals and the common porpoise. They all can accumulate considerable burdens of lipophilic compounds such as DDT as the final integrators of marine contamination. For these migrating animals, regional comparison is allowed on a very large scale only because the place where they were taken may not represent their normal place of residence.

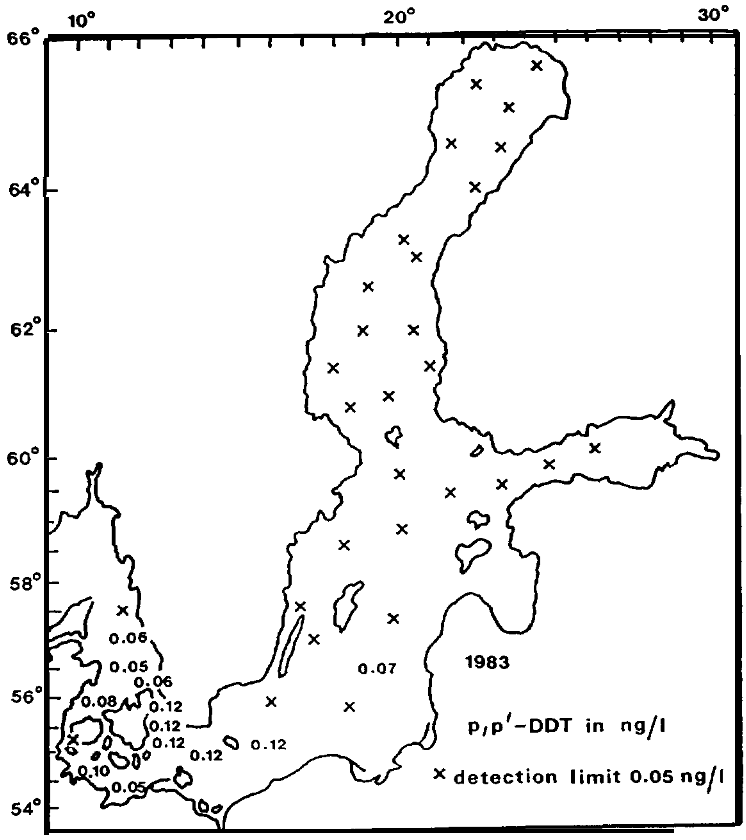

Figure 3. DDT-concentrations in water of the Baltic Sea in 1983.

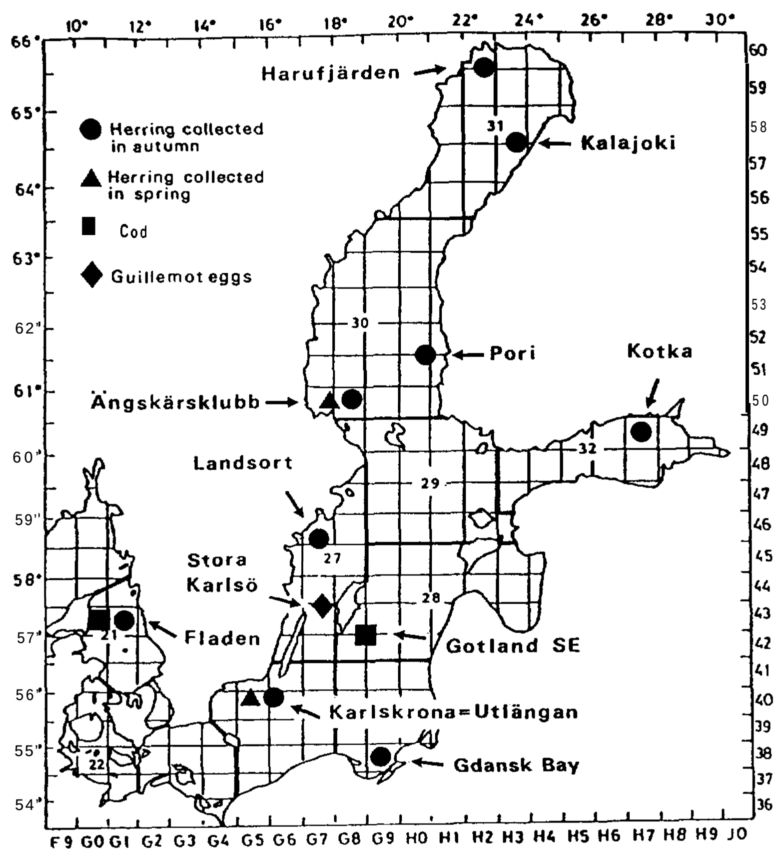

Figure 4. Collecting areas for harmful substances in biota.

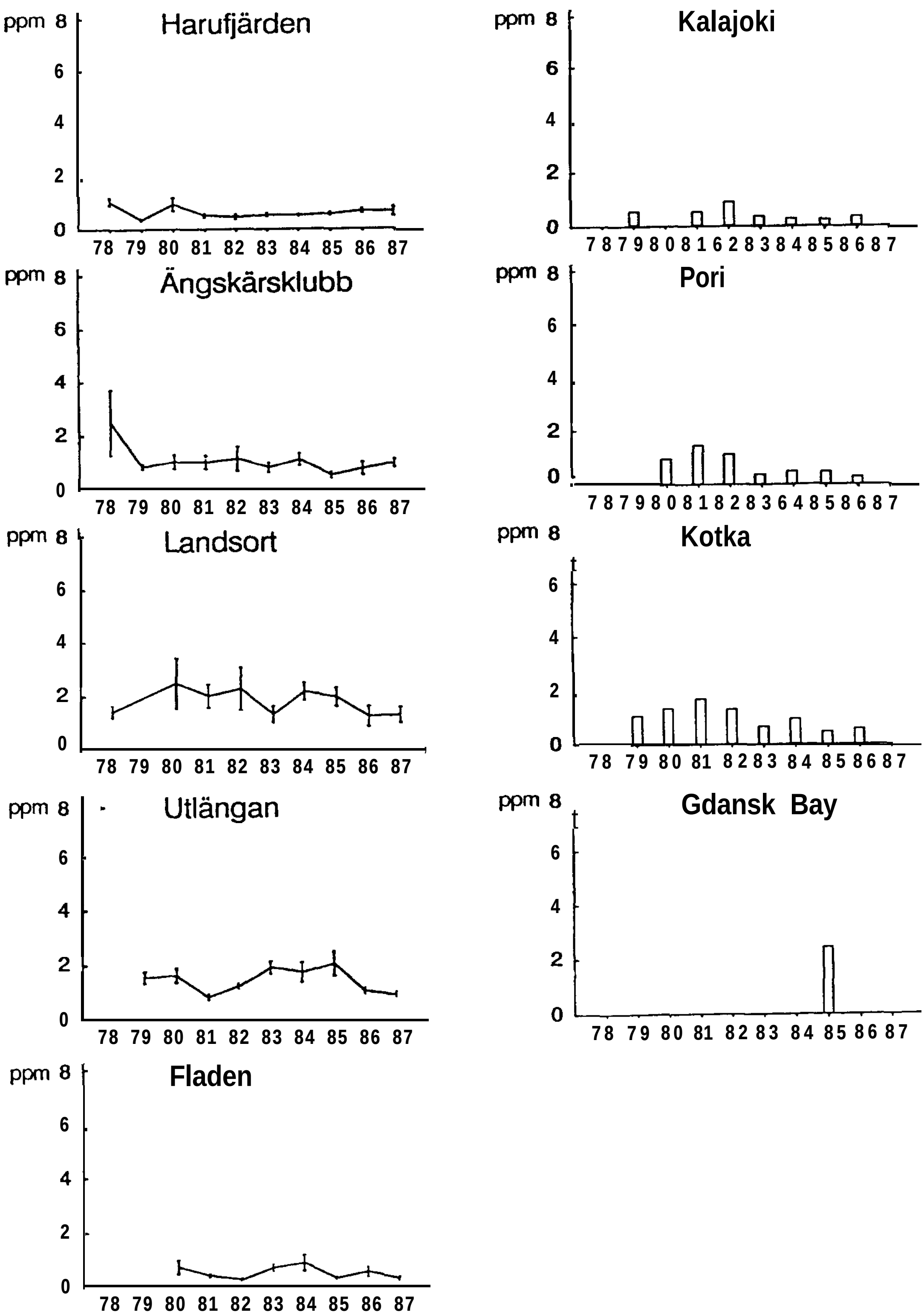

Figure 5. Annual vdriability and regional distribution of DDT in herring (from Odsjö & Olsson 1988, Haahti & Perttilä 1988, ICES 1988b).

However, there is too few available data to draw firm conclusions and often the normalising parameters as regards age, sex, or fat content are not reported. The ICES Working Group on Seals in the Baltic hopefully can give clarification on the specific burden of marine mammals in the Baltic compared with other sea areas.

9.3.2 HCHs

At present, Lindane (γ-hexachlorocyclohexane (γ-HCH)) can be detected in water of the entire Baltic Sea. Compared to DDT, it has the advantage of lower persistence (half-life: 2 years) and lower bioaccumulation (BCF: 200 to 500). It has been suggested as a tracer for large-scale hydrographic processes, e.g. the water exchange in the Kattegat (Briigmann et al. 1985).

In addition to Lindane, the isomers a-HCH and ß-HCH are found, which do not possess insecticidal properties. They originate from the time when technical HCH *) was used as an insecticide, and are still transported via the atmosphere from sources far away.

Since the Baltic Sea states discontinued the use of technical HCH between 1970 and 1980, a decline in its main component, a-HCH, has been ascertained (Figs. 6-7). This trend is clearly visible in a time series taken in the Arkona Basin (Gaul 1989).

The very even distribution of the HCH isomers which were ascertained in a close-mesh measurement network in 1983 (Gaul 1984) and 1987 (Gaul 1989), could be essentially confirmed in 1988. In the case of a-HCH, a decline was recognisable that was already clear in the time series (Fig. 8); for Lindane, a decline was is indicated in the conspicuous shifting of the 2.0 and 3.0 isopleths (Fig. 9); however, a confirmation of this development must still be awaited.

The concentrations of HCH isomers measured in the water represent no danger for the marine ecosystem according to the present state of knowledge, but they are indicators of an anthropogenic contamination of the Baltic Sea which would be avoidable if rapidly decomposing pesticides were used exclusively. In accordance with the low BCF values, the concentrations of HCH residues in biota are generally low (Tab. 6).

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*) Composition of technical HCH:

60 to 70 % a-HCH	7 % δ-HCH
7 to 10 % R-HCH	2 % ϵ-HCH
14 to 15 % τ-HCH	

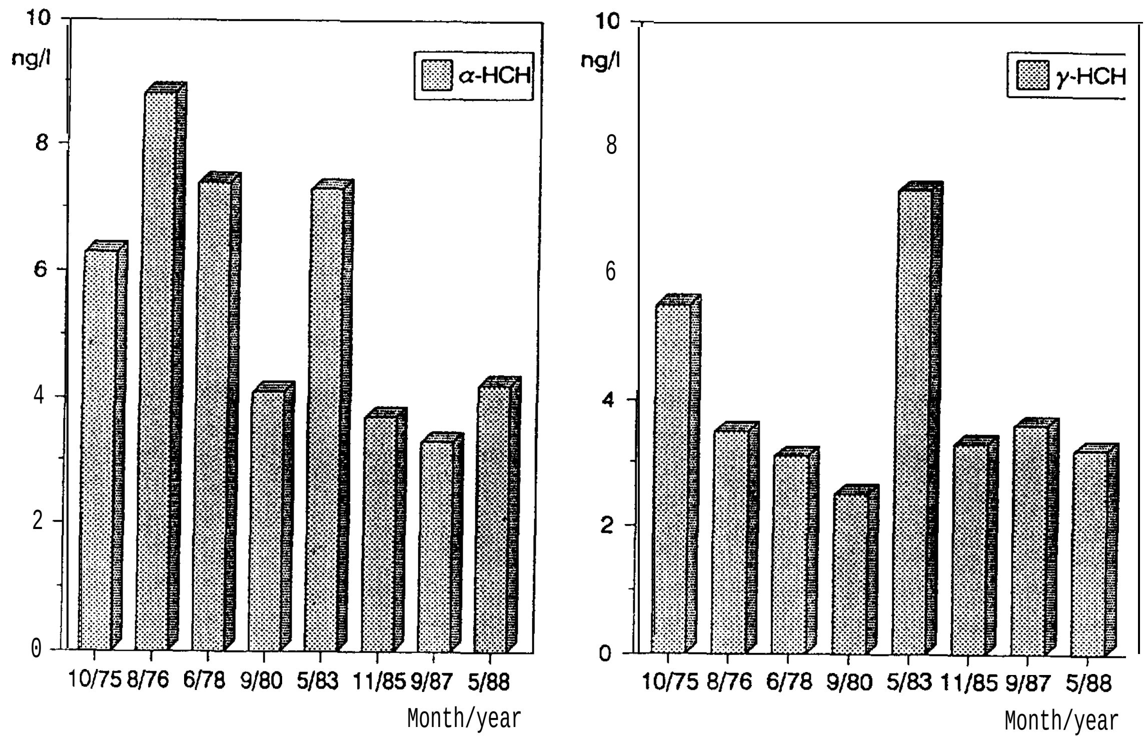

Figure 6. Concentrations of **HCHs** in surface water of **Kiel** Bay (N 3).

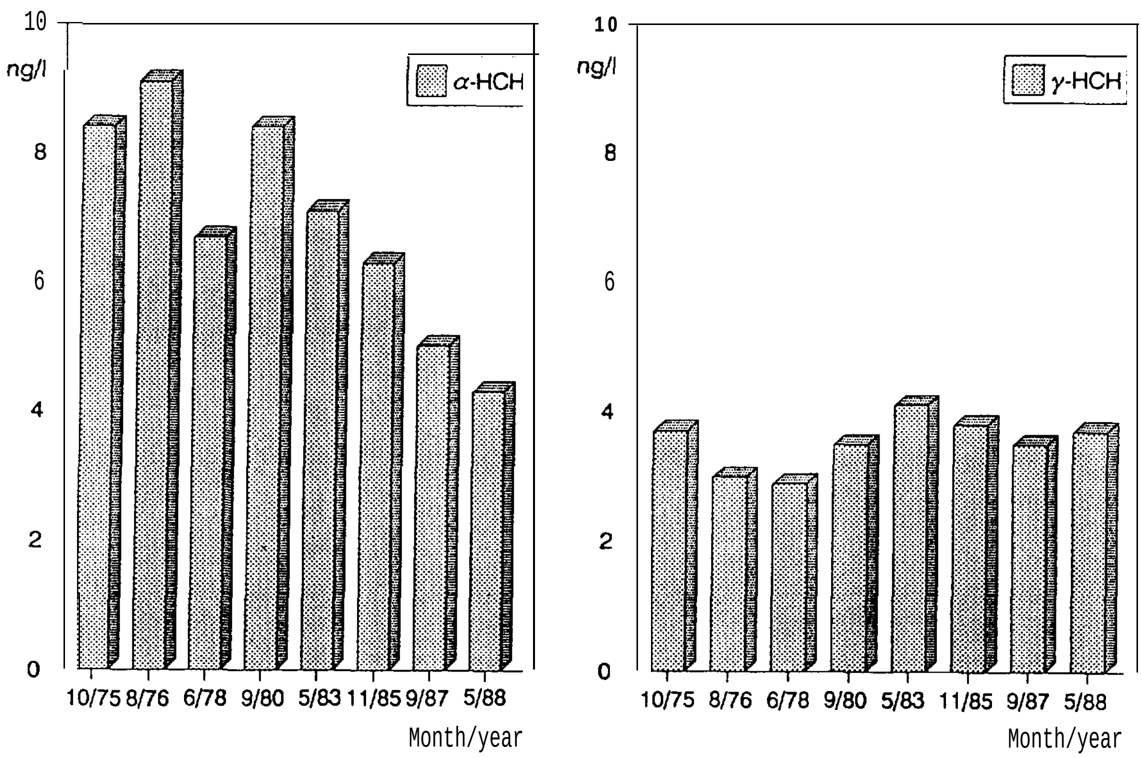

Figure 7. Concentrations of **HCHs** in surface water of Arkona Basin **(K 4).**

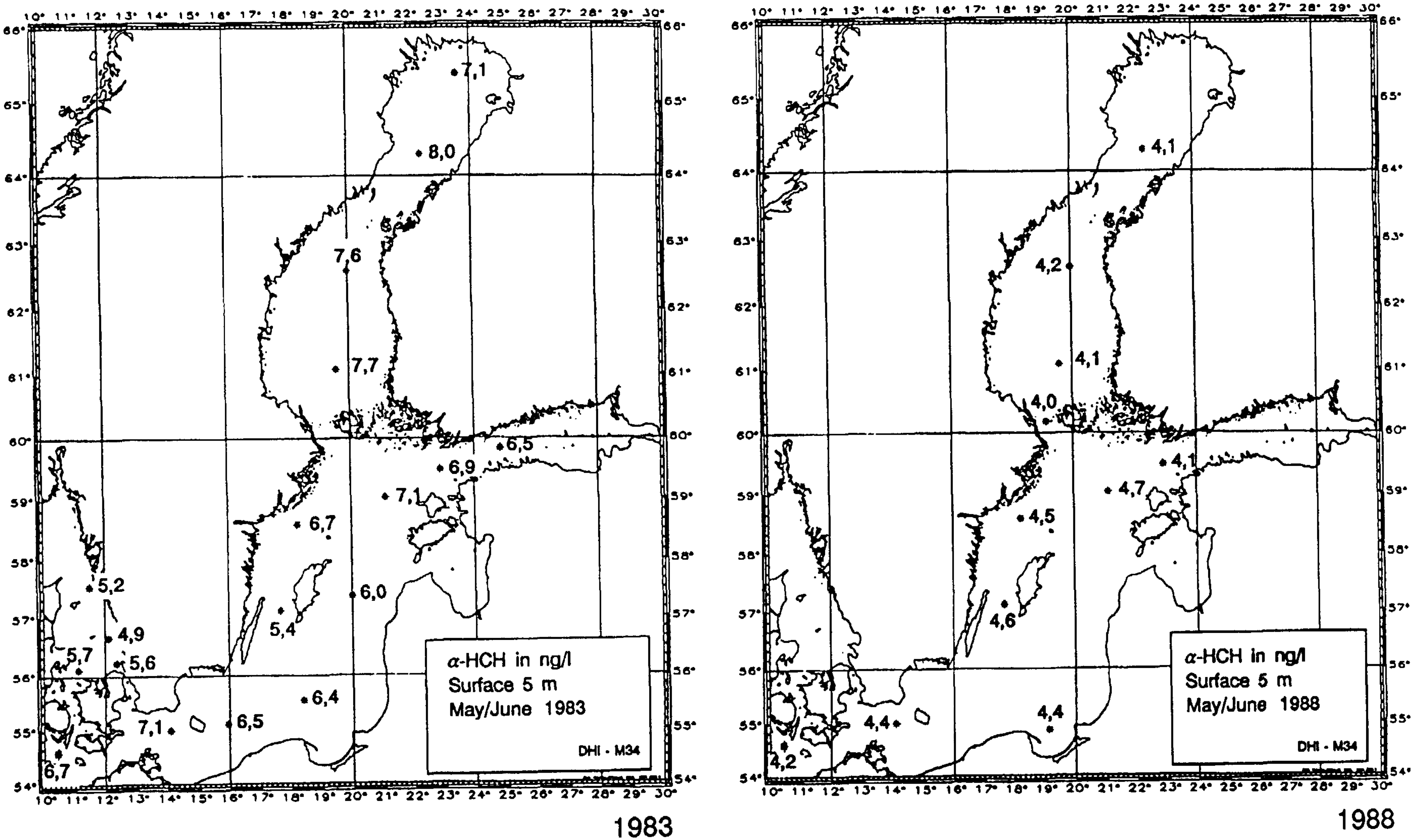

Figure 8. Surface water concentration of α-HCH in 1983 and 1988.

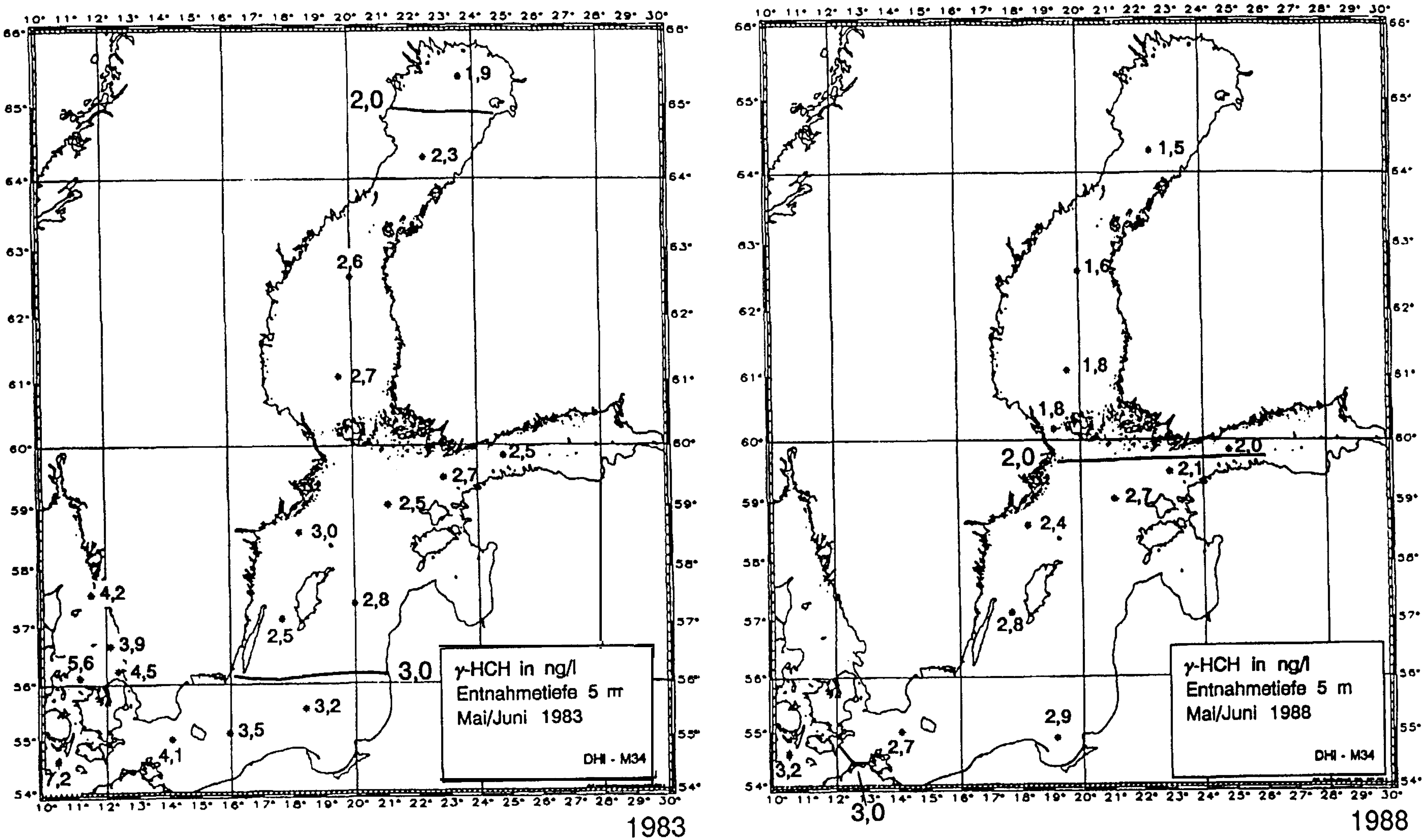

Figure 9. Surface water concentration of Lindane (γ-HCH) in 1983 and 1988.

9.3.3 Chlorinated camphenes (PCC)

Chlorinated camphene is used as a pesticide in countries around the southern Baltic Sea. The commercial names of the products are: Toxaphene, Strobane, Melipax, or Phenacide. They represent complex mixtures of substances with up to 8,500 individual components. They elute together with the PCBs during gas chromatographic investigations if they are not separated beforehand by sophisticated methods.

PCC residues in organisms can amount to up to 10 % of the respective amount of PCBs, and thereby are in the same order of magnitude as DDT. However, the regional differences between the Kattegat and the central Baltic Sea are less pronounced than is the case with DDT (Bernes 1988) due to the fact that the input of PCCs into the Baltic Sea by atmospheric deposition is comparable to, or even larger than, the input by run-off from land. This leads to the striking result that PCC levels in seals and marine birds from Spitzbergen are almost equal to those in the corresponding biota in the Baltic Sea, though the DDT and PCB concentrations differ considerably (Andersson et al. 1988c). The quantitative determination is made difficult because the PCC pattern is changed by decaying processes compared with the initial mixture (Zell & Ballschmiter 1980).

PCC residue concentrations in fish from the Baltic Sea are higher than those of DDT; in the case of seals and sea gulls, the ratio is reversed, which is indicative of a faster metabolism of PCC in warm blooded animals (Widequist et al. 1984).

From the example of PCC, it is clear that substances should not be deliberately discharged into the environment if an adequate possibility does not exist to assess their distribution with justifiable analytical expense. Such an assessment, for more than 8,000 components as in the case of PCCs, cannot be carried out.

9.3.4 Other chlorinated pesticides

Of the rest of the chlorinated pesticides mentioned, - Dieldrin, Chlordane, Heptachlor, and Mirex, concentrations were found in water which were below or only slightly above the limits of detection of the method used. Their occurrence in organisms is reported occasionally, but their quantities do not exceed 10 % of the Σ DDT concentration. Substantiated indications of negative effects upon the organisms in the Baltic Sea are not available (Bernes 1988).

9.3.5 Sediments

Many substances are adsorbed onto fine grained organic detritus, are then transported to sedimentation areas and there trapped in marine sediments. For organic compounds under aerobic conditions, microbial decay takes place. Under anoxic conditions, chemical changes may occur (e.g. DDT to DDD), however, a complete mineralization is extremely slow.

In order to compare the toxic substance concentrations measured in different types of sediment, a normalization to the content of organic carbon (TOC) in the sediment is necessary (Andrulewicz et al. 1979, Lohse 1988). This normalisation of the data has not been done thus far. Moreover, the comparability of the data has not been ensured because an intercomparison of analyses has not yet taken place. The concentrations reported thus far (Perttilä & Haahti 1986) for lipophilic substances in the sediments are several magnitudes higher than those in water. However, in the evaluation one must take into account that the bioavailability of material adsorbed in the sediment is several orders of magnitude smaller than compared with substances dissolved in water (Neff 1984).

9.4 PCBs AND "NEW CONTAMINANTS"

O. Svanberg[1] and M. Olsson[3]

The group of pollutants covered by the term "New contaminants" in this report includes those compounds listed as "New contaminants" in document WGS 6/5 of the Ad hoc Working Group on Criteria and Standards for Discharges of Harmful Substances into the Baltic Sea (Baltic Marine Environment Protection Commission - Helsinki Commission 1983), the "old" polychlorinated biphenyls (PCBs) and some additional compounds more recently discovered as potentially harmful. It has been found appropriate to separate this group of contaminants from pesticides intentionally released in the environment when used for pest control. The "New contaminants" covered below are?

- polybrominated biphenyls (PBBs)
- polybrominated diphenylethers (PBDEs)
- polychlorinated terphenyls (PCTs)
- polychlorinated naphthalenes (PCNs)
- polychlorinated paraffins (CPs)
- polychlorinated benzenes (PCBz)
- polychlorinated phenolic compounds (PCPs)
- polychlorinated dibenzo-p-dioxins (PCDDs)
- polychlorinated dibenzofurans (PCDFs)
- phthalic acid esters (PAEs)
- nonylphenols (NPs)
- halogenated organic material analysed as total organic bound chlorine (TOCl), extractable organic bound chlorine (EOCl) or adsorbable organic bound halogen (AOX).

Organo-tin pollution of the sea originating from the use as an antifouling agent is discussed in Chapter 8 "Trace Elements".

Of the listed contaminants above only PCBs are obligatory in the second stage of Baltic Monitoring Programme.

There are several additional candidates of potential contaminants to be covered by this report but on which no reliable data on the occurrence in the environment exist. It is therefore a recommendation to consider these in the future search for environmental contaminants to be considered for regular monitoring. Examples of this are:

polychlorinated thiophenes
polychlorinated diphenylethers
modified resin acids.

9.4.1 Polychlorinated biphenyls

Polychlorinated biphenyls (PCBs) belong to the group of persistent bioaccumulating pollutants which have seriously contaminated Baltic waters. They were mainly used in industrial products and even if their use has been banned or strictly regulated since the beginning or middle part of the 1970s, their presence in various products still in use implies that leakage to the environment still occurs. This stresses the importance of trend monitoring studies.

Baseline and monitoring studies

A number of national reports concerning changes over time in concentrations of PCB in various organisms have been presented during recent years. When comparing data from different localities and years, it is important to know as many parameters as possible which explain the variation found in a sample. Perttilä et al. (1982) have found an age dependence of organochlorines in Baltic herring. This has earlier been found by Jensen et al. (1972) in Baltic herring. In a recent report, dioxins also show an age dependence (Bergqvist et al. 1989b). This confirms the importance of selecting specimens of similar age for trend monitoring studies.

Trend monitoring

Some reports concerning changes in levels over time covering a period of more than a few years have been presented. Haahti and Perttilä (1988) have studied one annual homogenate of 20 herrings collected in autumn from four localities along the Finnish coast during the period 1979-1986 and an unknown number of individual cod livers collected annually in autumn during the period 1980-1985 from two localities (Fig 4). Their data show a decrease in PCB concentrations over time in all areas for both species (Fig. 10).

PCB studies in Swedish waters comprise several different trend studies. All series have been analysed at the same laboratory using the same analytical procedure. This implies that the data are comparable.

Herring have been collected in spring annually during the period 1970-1987 at two localities. The annual samples consist of 20 individuals (Olsson and Reutergårdh 1986, Odsjij and Olsson 1989). Herring has also been collected annually *in* autumn at five localities and 20 specimens have been analyzed individually. The series covers the period 1978-1986 (Odsjij and Olsson 1988). Finally, 10 eggs of guillemot have been analyzed annually during the period 1967-1987 (Olsson and Reutergårdh 1986, Odsjö and Olsson 1989).

Cod liver has been sampled annually in autumn at two localities during the period 1980-1986. The annual samples consist of 20 specimens and they have been analysed individually. All sampling localities are presented in Figure 4.

The series covering the longer periods, e.g., spring samples of herring and guillemot eggs from the Baltic Proper, show significant decreases in PCB concentrations over time, most of which occurred during the middle part of the 1970s (Fig. 11 and Fig. 12). The annual samples collected in spring in the southern Gulf of Bothnia do not show a significant decrease (Fig. 13). Neither does the series of autumn collected herring show any significant change over time at the five localities during the investigated time 1978-1986 (Fig. 14). The cod series might indicate a slight decrease over time (Fig. 15) (Odsjb and Olsson 1988).

Finnish scientists have investigated tissues from grey seals, consisting of animals collected during their first year of life. The material comprises 28 specimens collected during the period 1981-1986. A trend analysis does not show any significant decrease in levels of PCB (Stenman et al. 1987). Unpublished material on 24 grey seals of 5-8 months age collected during 1974-1976 and on 16 animals of the same age collected in 1982-1986 have been analysed by Swedish scientists. The samples were collected in the Gulf of Bothnia. The Swedish data revealed *no* statistically significant difference in levels of PCB (Fig. 16) (Olsson and Reutergårdh, unpublished results, see also Odsjö and Olsson 1989). In addition, 15 grey seals of the same age, but collected in the Baltic Proper during the 1980s, were analysed. No difference in levels between the Gulf of Bothnia and the Baltic Proper was found during the 1980s. Grey seals are known to migrate between the Baltic Proper and the Gulf of Bothnia.

Perttilä and Haahti (1986) have studied PCB levels in sediment samples collected in different areas in the Gulfs of Bothnia and Finland and in the Baltic Proper. The levels are higher in the Gulf of Finland and the Baltic Proper. The sediment cores were cut into 1 cm thick slices and analyses normally show increasing levels from deeper layers up to the surface. The authors interpret the findings such that the sediment layers reflect the situation in the water mass at the time of deposition. In the sample from the station in the Bothnian Bay, however, much higher levels were found in the intermediate layers 5 to 11 centimeters deep. These data seem anomalous since the authors state that the sedimentation rate at this station is 1 cm per 8.3 years. This would imply a maximum contamination of the area 90-50 years ago, which cannot be true. Since the DDT levels investigated in the same core do not increase in the intermediate part of the core, bioturbation does not seem to be the explanation.

In a study of human breast milk in Sweden, Norén (1987) found a decrease in PCB levels. The material covered the period between 1972 and 1985. The values decreased from 1.05 to 0.60 $\mu g/g$ lipid weight. It is not known to what extent the decrease is caused by lower PCB levels in fish or to the fact that Swedish authorities have given recommendations concerning the diet during pregnancy, most probably implying a lowered fish intake among pregnant women.

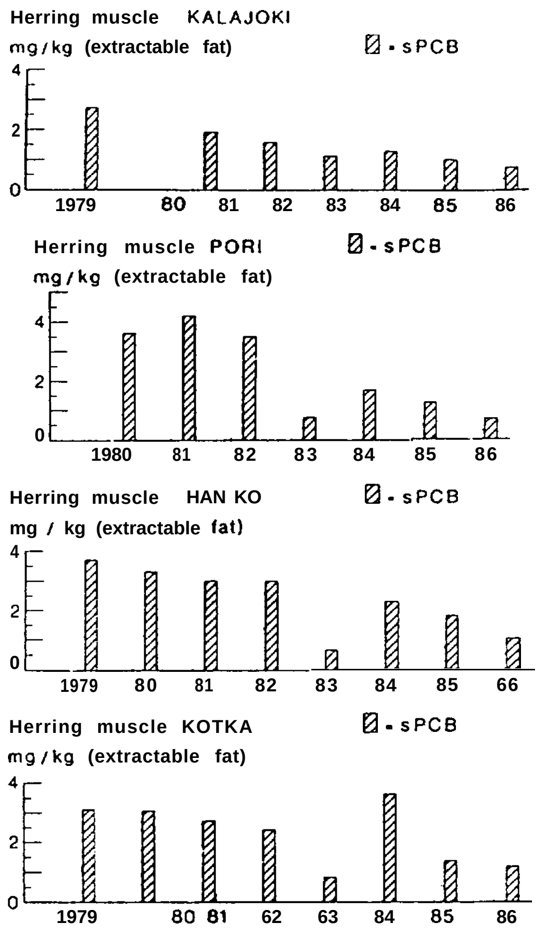

Figure 10. Levels of PCB (lipid weight basis) in muscle of herring annually collected in autumn at the Finnish coast (from Haahti & **Perttilä** 1988).

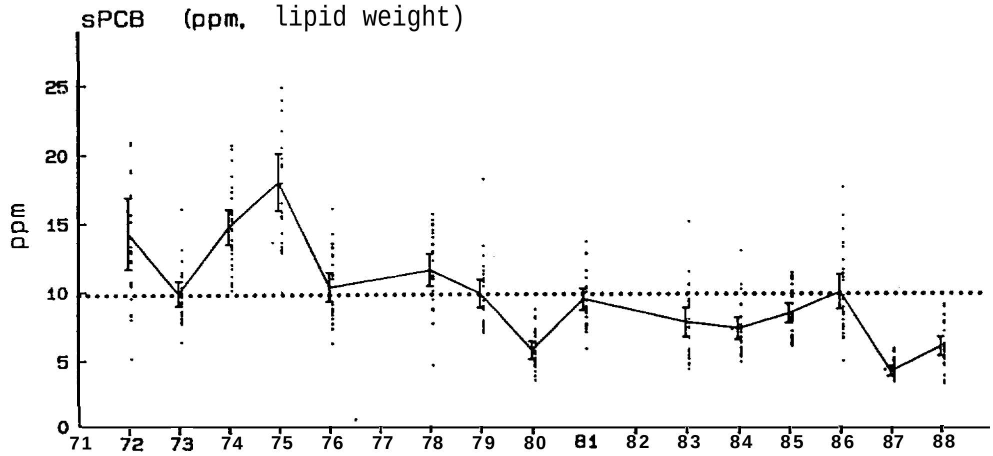

Figure 11. Levels of PCB in muscle (lipid weight basis) of herring annually collected in spring from outer Karlskrona archipelago (Baltic Proper).

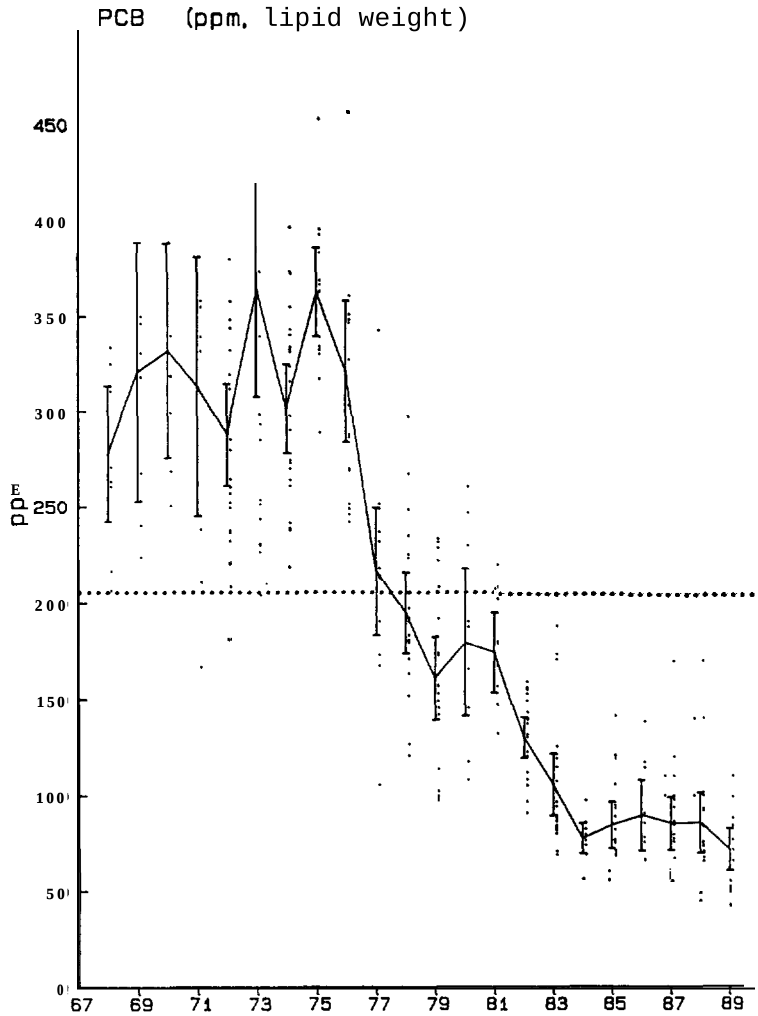

Figure 12. Levels of PCB (lipid weight basis) in annual samples of guillemot eggs annually collected from Stora Karlsö (Baltic Proper).

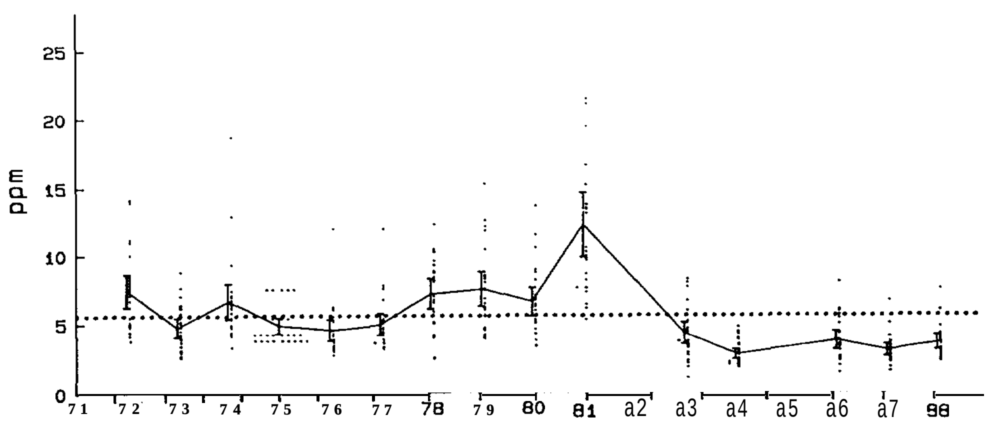

Figure 13. Levels of PCB (lipid weight basis) in muscle of herring annually collected in spring from Ängskärsklubb, southern Bothnian Sea.

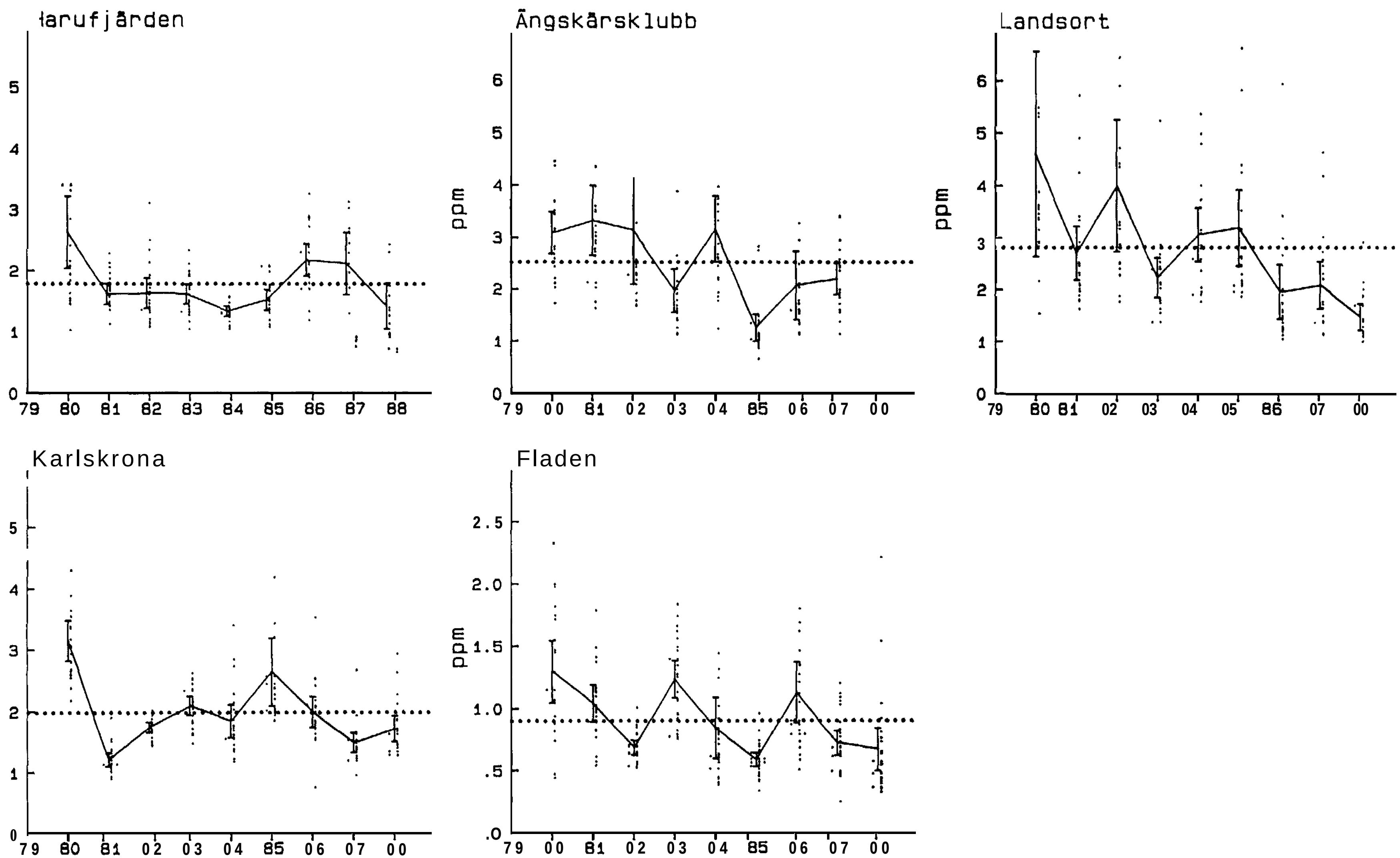

Figure 14. Levels of PCB (lipid weight basis) in muscle of herring annually collected in autumn at five different sites.

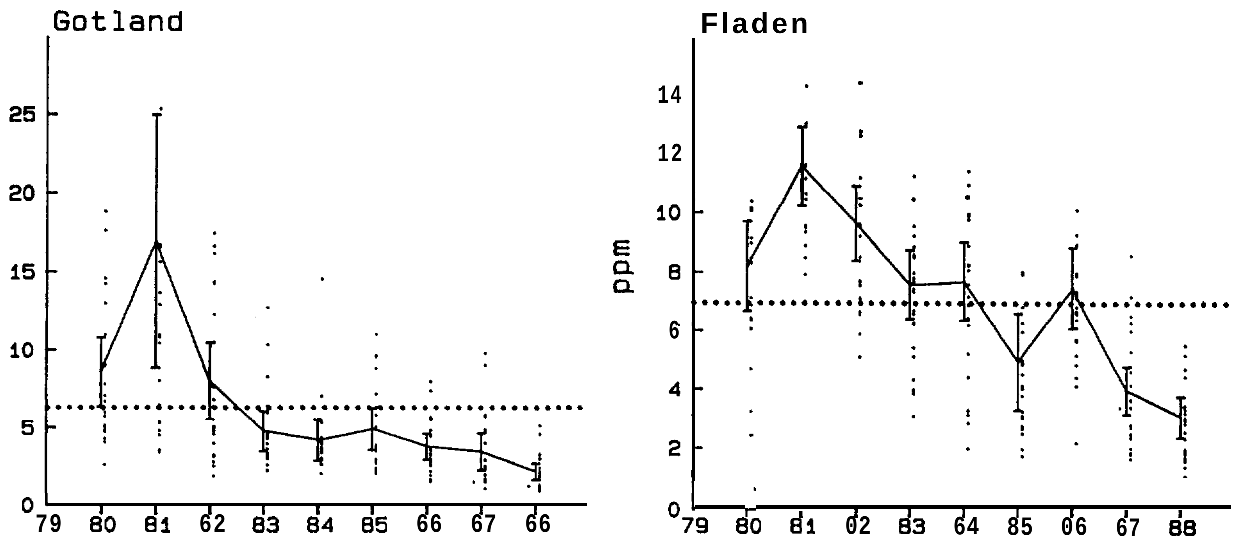

Figure 15. Levels of PCB (lipid weight basis) in liver from cod annually collected during autumn from the Central Baltic (Gotland) and from the Kattegat (Fladen).

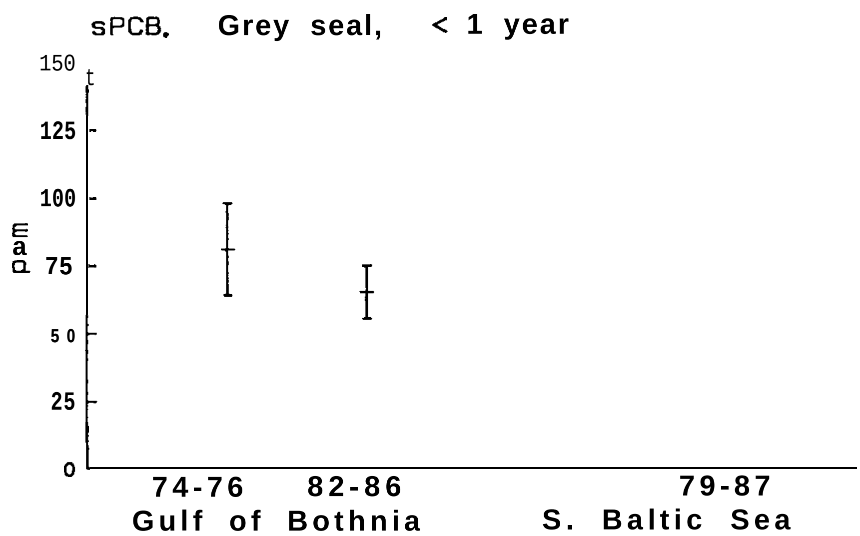

Figure 16. Levels of PCB (lipid weight basis) in blubber from yearlings, of grey seal, 3-8 months old, and with normal weights. The samples were collected from the Gulf of Bothnia and the Southern Baltic Proper during two periods.

From available data, it is obvious that most of the changes over time in PCB levels occurred during the 1970s. This decrease was probably caused by the governmental bans on PCB in Europe. Assuming that the production and any new uses of PCB have stopped, the only small changes during the 1980s found by Finnish investigators and the lack of significant changes found in Swedish waters might imply that products still in use continue to leak PCB to the environment. If we want to improve the conditions in the environment, the search for old sources and PCB contaminated products have to be more intensive and PCBs have to be destroyed under controlled conditions.

Spatial variation

The mean values of PCB for the entire period investigated at the five sampling localities where herring have been collected during autumn show significant differences (Fig.17) (Odsjö and Olsson 1988). The highest values are recorded in the northern part of the Baltic Proper, whereas lower levels are found in the Bay of Bothnia and even lower at the Swedish west coast. It is interesting that cod liver from the Swedish west coast have similar levels to Baltic cod liver (Fig. 18).

Available herring data from the Finnish investigation (Haahti and Perttilä 1988) show lower PCB levels in the samples from the Gulf of Bothnia than in the samples collected in the Gulf of Finland. The PCB levels are slightly lower than the data obtained on the Swedish side. Whether this is due to differences in the analytical procedure or differences in the pollution burden between the western and eastern part of the Gulf of Bothnia is not known. Both samples consist of young herring collected during autumn.

PCB congeners

Recently, investigations have focused on the composition of the individual chlorobiphenyl (CB) congeners present in biota. The analytical technique has improved and the use of capillary columns have made it possible to analyze some individual congeners. Within the PCB group, much attention has been paid to coplanar molecules. Within this group, CBs having similarities in toxicity to dioxins are found. Depending on structural differences between the various congeners, different CBs might bioaccumulate and metabolize at different rates. In particular, coplanar molecules might differ from others.

Baltic herring muscle and cod liver have been analyzed for some of the different PCB congeners by Haahti and Perttilä (1988) and Roots (1989). The relative proportion of CBs analysed in these two studies are in good agreement. This might indicate that there is no obvious variation in the pattern of the CB congeners studied in different Baltic areas as long as the same species is concerned. Haahti and Perttilä have studied one herring sample from the Gulf of Bothnia and one cod liver sample from the Baltic Proper. Roots has studied three samples of herring from the Gulf of Finland and the Baltic Proper. A comparison between the CB patterns for the herring sample from the Gulf of Bothnia and a cod liver sample from the Baltic Proper might indicate a minor species or organ difference.

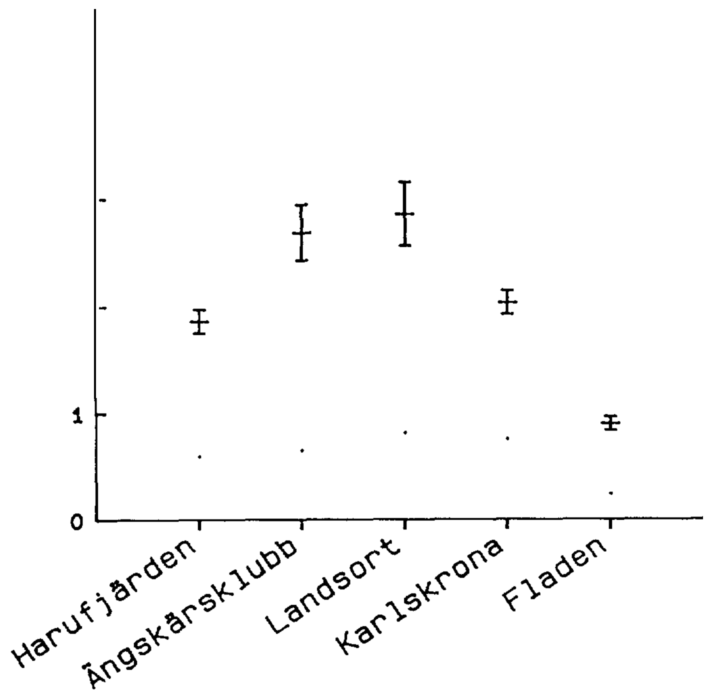

Figure 17. Mean levels of PCB (lipid weight basis) for the period 1978-1988 in herring collected in autumn at five different sites.

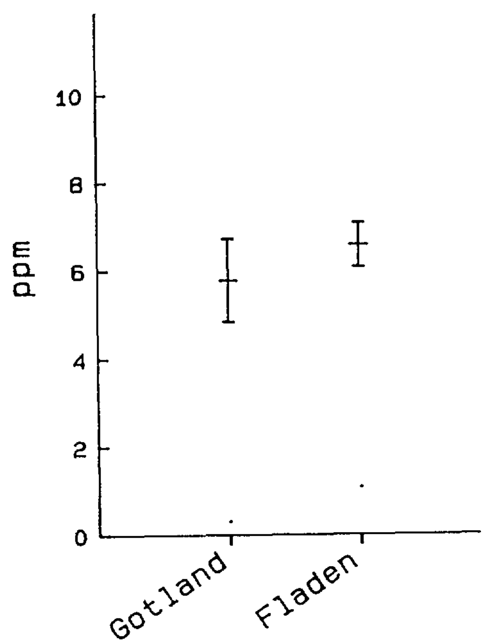

Figure 18. Mean levels of PCB (lipid weight basis) for the period 1978-1988 in cod collected in autumn in the Baltic Proper (Gotland) and in the Kattegat (Fladen).

Levels of coplanar CB congeners in Baltic biota (salmon, white tailed sea eagle) have been presented in several reports (Tarhanen et al. 1988, Koistinen et al. 1989, Paasivirta et al. 1989). The coplanar CB congeners are known to induce liver enzymes in a way similar to dioxins and dibenzofurans. This implies that their toxicity can be expressed in so called TCDD equivalents (see also sub-chaper 9.49). Tarhanen et al. (1988) found that the presence of coplanar CBs contributed more to the presence of TCDD-equivalents in Baltic sea eagles than TCDD (tetrachlorodioxins) and other dioxins and dibenzofurans themselves.

Effects on mammals

In recent literature, PCB induced effects on Baltic organisms have mostly dealt with effects on mammals. A disease complex most probably occasioned by a primary lesion on the adrenal cortex (adrenocortical hyperplasia) causing secondary lesions in various other organs has been reported among Baltic seals . The pathological changes found in the animals were skin lesions, digital lesions, periodontitis, severe bone erosions of the jaws, regional intestinal ulcers, arteriosclerosis, adrenocortical hyperplasia, renal glomerulopathy, renal tubular hyperplasia, uterine occlusions and leiomyoma. Reproduction was impaired.'Organochlorines and especially PCBs were believed to be the primary cause of the disease complex (Bergman and Olsson 1986 and 1989, Bergman et al. 1989, ICES 1988a).

The population sizes of all three seal species, grey, common and ringed seal, are today only a fraction of the size at the beginning of this century (Almkvist 1980, Durant and Harwood 1986, Helander 1989a,b, ICES 1984, 1987). The ringed seal population seems to have decreased since the 1970s (Helle 1986), but there is no indication of a further decrease in the entire Baltic grey seal population during the last 12 years (Helander 1989a,b). There are, however, regional differences so that the number of grey seals in the Gulf of Bothnia seems to have increased slightly and the number in Baltic Proper seems to have decreased.

Helle and Stenman (1987) reported signs of an increased reproductive rate during the 1980s among ringed seals in the Gulf of Bothnia. The signs seem encouraging, but so far no increase of the population has been seen (Härkönen & Heidi-Jorgensen 1989a).

The epizootic caused by PDV (Phocine Distemper Virus) during summer 1988 induced a 60% mortality of the common seal populations in Skagerrak, Kattegat and southwestern part of the Baltic (Härkönen and Heide-Jørgensen 1989b, Helander 1989b). So far there are no evidence associating environmental contamination with the epizootic even though it cannot be excluded that a decreased immune defense caused by pollution might have given the epizootic disease a more severe course (ICES 1989). There are no indications today that seals in Kattegat and Skagerrak have a lowered immune defense, but this has not been carefully investigated.

The Baltic otter population has almost disappeared during the last decades. As in the case of the seals, this happened during a period when the organochlorines became a severe environmental problem. Since there are no otters left to be investigated in Baltic archipelagoes, the cause of the population decrease cannot be investigated. It is, however, known that otters collected in Swedish fresh waters have had high levels of PCB (Sandegren et al. 1980). The levels are as high as those that cause reproductive impairment in laboratory investigations on mink. Some few otters collected in Swedish archipelagoes during the end of the 1960s and beginning of the 1970s also had very high PCB levels (Olsson et al. 1981). In all Sweden, the otter population has decreased severely during the last decades. The present otter populations in fresh waters of southern Sweden seem only to have survived in some few highly eutrophicated waters (Olsson and Sandegren 1989a,b). In northern Sweden, a similar pattern is also seen and otters are mostly found in areas with high ground quality with regard to nutrients and in areas downstream of water sewage plants (Olsson et al. 1989). The high PCB levels found also in otters from remote areas in Sweden far away from local PCB discharges and the present distribution pattern for the surviving otter populations indicate that eutrophication in water has had a positive effect on fish consumers. The implication of eutrophication on levels of bioaccumulating substances is also discussed in the introduction to Chapter 9.

9.4.2 Polybrominated biphenyls

Polybrominated biphenyls (PBBs) are used as flame retardants. They are persistent and bioaccumulating compounds. The toxic properties are similar to those of PCBs.

Environmental levels

Levels of PBB in seals and guillemots collected from the Baltic Sea are two to five times higher than in the same species from the North Sea and from the Arctic Ocean (Jansson et al. 1987). Values reported for common seal, guillemot and sea eagle from the Baltic were 20, 160 and 280 ng/g lipid, respectively.

9.4.3 Polybrominated diphenyl ethers

Polybrominated diphenyl ethers (PBDEs) are used as flame retardants. They are distributed in the environment from treated products and from their use in textile industry.

Environmental levels

Jansson et al. (1987) report levels in common seal, guillemot and sea eagle at 90, 370 and 350 ng/g lipid.

Recent studies on stratified sediment from the Bornholm area show PBDE levels which indicate an increase by up to 20 times during the last decades (Sellström et al. 1989). The sediment PCB levels during the same period are fairly unchanged.

9.4.4 Polychlorinated terphenyls

Polychlorinated terphenyls (PCTs) have been industrially produced and used for the same purposes as **PCBs.** The global distribution indicates long-range transport by air. There are **several thousands** of theoretically possible **congeners,** making up a very complex mixture in the technical product.

Environmental levels

No data have been found reported after the findings in late 1970s of PCT in Baltic Sea mammals and birds and referred to in the first periodic assessment (Slaczka et al. 1987).

These findings were based on the method of perchlorination and analysis with gas chromatography on packed columns, both not very reliable from the present state of knowledge. The scientific community is requested to confirm the previous findings by more reliable methods.

9.4.5 Polychlorinated naphthalenes

Polychlorinated naphthalenes (PCNs) as a technical product are used as a flame retardant, **as** insulating material in capacitors and as a biocide, e.g., in wood preservatives. The world production is considered to be decreasing. **PCNs** are also unintentionally formed in incineration processes and in the production of magnesium.

The use and **release pattern into the environment resembles** that of PCBs **and PCNs are** also regarded as a global pollutant.

PCN is only slowly degraded in the environment and is bioaccumulating.

Toxicity increases with increasing number of chlorine atoms in the molecule.

Environmental levels

Only **a** few studies on PCN levels in environmental samples have been reported and the small number of samples and specimens analysed make the results only indicative.

A brief survey (Jansson et al. 1984) on organisms from different parts of Sweden show PCN levels between 3 and 62 ng/g (lipid basis). The lowest value was found in char from a "clean" mountain lake, while the highest value was from liver of cod caught in the outer Stockholm archipelago. Freshwater'pike, Baltic salmon, white-tailed eagle and grey seal all show intermediate levels.

Tarhanen et al. (1988) report somewhat higher levels in Baltic salmon (70 ng/g) and in white-tailed eagle (200 ng/g).

One specific component of the complex mixture in the technical product is retained in organisms (Asplund et al. 1986), which explains the peculiar chromatographic peak pattern seen from environmental samples.

9.4.6 Chlorinated paraffins

Polychlorinated paraffins (CPs) also named polychlorinated alkanes (PCAs) constitute a mixture of a large number of congeners. The carbon chain length varies between 10 and 30 and the degree of chlorination varies from 40 to 70%. For a review, see Svanberg (1983) .

CPs are used mainly as flame retardants, plasticisers and as additives in cutting and drilling fluids used in metal works.

Experimental studies have revealed that CPs bioaccumulate in fish and that for certain types of CPs the elimination rate from fish is extremely slow.

The acute toxicity to mammals and fish is low, but small-sized crustaceans are acutely affected at ng/g levels in the surrounding water. Long term exposure of fish causes signs of reversible neuro-toxic effects.

Levels in the environment

Due to analytical difficulties, there are few investigations of CPs in the marine environment reported.

Wideqvist et al. (1989), in a comparative study of CPs in biological samples, discovered a distribution pattern in the environment quite different from that of PCBs. The highest concentrations were found in rabbit and moose from southern Sweden (3 and 4.5 mg CP/kg extractable lipid respectively) and lower levels in grey seal from the Baltic and in osprey (less than 1 mg/kg). Herring from the Bothnian Sea, from the Baltic Proper and from the Skagerrak all have rather similar levels, at l-2 mg/kg. The lowest levels were found in reindeer from northern Sweden and ringed seal from Spitzbergen.

9.4.7 Polychlorinated benzenes

Hexachlorobenzene (HCB) and other chlorinated benzenes (CBz) are unintentionally formed in several industrial processes, e.g. magnesium production, chlorine gas production from sea water, production of chlorinated solvents. HCB is also formed in waste incineration. The use of HCB as a fungicide starting during the 1940s ceased in the western countries around 1975.

Due to the high volatility, the emitted product is to a great extent transported by air and thus globally distributed.

The chemical properties of HCB result in a poor degradation in the environment.

The accumulation of the chlorinated, benzenes in biota and sediment increases with increasing degree of chlorination.

The acute toxicity of HCB is low, but long-term exposure to low concentrations might give rise to sub-lethal responses in aquatic organisms.

A review and hazard assessment of HCB has been published recently (Wachtmeister & Ekelund 1989).

Environmental levels

Water

The levels of HCB in Baltic Sea water sampled in May-June 1983 were 10-40 pg/l without any clear regional differences (Gaul 1984). In remote areas of the North Sea, Gaul and Ziebarth (1983) reported around 20 pg/l while Ernst (1986) recorded 30 pg/l from the outer parts of the German Bight and 3 000 - 15 000 pg/l in the Elbe Estuary. This can be compared with values of 3-22 pg HCB/l in surface waters of the Atlantic Ocean (Krämer and Ballschmiter, 1988).

Sediment

The content of HCB in sediments from Elbe estuary was in the range 15 - 40 ng/g dry weight (Ernst 1986), while in heavily contaminated areas in Frierfjord in Norway, the corresponding values were 320 - 530 ng/g (Bjerk & Brevik 1980).

Biota

The Baltic HCB contamination is mirrored by levels in herring at ca 100 ng/g (lipid base) measured in 1989. This is about half of the levels seen at the beginning of the decade. The corresponding levels in the North Sea are about one tenth lower. Herring sampled during the years 1981-1989 revealed a range of 50-430 ng/g (lipid base)depending on season, year and sampling site. No significant trends in the levels were observed after 1983 (Reutergårdh, pers comm).

Also Krüger & Kruse (1989) report HCB levels in herring from the Bornholm area about ten times higher than in herring from the Shetlands (Table 6).

Perttilä et al. (1982) in a study on age dependent HCB-levels in herring found no such significant correlation. The levels in the fish samples (wet weight basis) varied between 0.56 and 1.75 μg/kg.

Sprat from the Polish fishing area of the Baltic Sea contained 13.4 μg HCB/kg (wet weight) compared to 1.1 μg/kg for cod, 5.9 for herring and 2.5 for flounder (Andrulewicz et al. 1988).

Analyses of HCB in ***Mytilus*** and plaice liver from Danish waters show generally low levels. The highest value for Mytilus, 0.74 ng/g (wet weight), was obtained from Rönne, Bornholm (Granby 1987)

Salmon caught in the open sea of the Baltic Proper and Gulf of Bothnia showed higher concentrations of HCB in extractable fat than salmon from the Gulf of Finland (Vuorinen & Paasivirta 1988). Temporal variations, however, make a strict comparison difficult.

A pooled sample of blubber from grey seals collected in 1979-1985 in the Baltic Sea contained around 200 ng/g HCB, thus, at the same level as in herring for the same period (Reutergårdh, pers comm).

HCB in guillemot (*Uria aalge*) eggs from Stora Karlsö has decreased to a steady level of 3-4 mg/kg (lipid base) after 1983 (Reutergårdh, pers. comm.), which is only a little lower than in this same bird species from Svalbard (Edelstam et al. 1980).

9.4.8 Chlorophenols

Chlorophenols, due to general toxic properties affecting the energy metabolism of the living cell, have been widely used as fungicides. This application is now restricted - in some countries banned, and the release into the environment is thus reduced.

Chlorophenolic compounds as a major group are also formed and discharged from the chlorine bleaching process of pulp mills. Formation results from a reaction with lignin residues.

The relatively high persistence and tendency to bioaccumulate in combination with the toxic properties of chlorophenols give rise to environmental concern (Renberg 1981).

Biotic and abiotic factors determining the fate of discharged chlorinated phenols in the aquatic environment have been studied by Xie et al. (1986). Phenols, guaiacols and catechols are transformed to veratrols and anisols by bacterially mediated methylation (Neilson et al. 1989). These metabolites are considered to be still more resistant to further degradation and the toxicity and potential for bioaccumulation are higher than the parent substances.

A tentative advanced environment hazard assessment of 4,5,6-trichloroguiacol has been performed by Neilson et al. (1989).

Emissions

Di-, tri-, tetra- and penta-chlorophenols together with di-, tri- and tetra-chloroguaiacols and chlorocatechols are all identified in pulp mill chlorine bleachery effluents (Xie et al. 1986). Chlorophenols and chloroguaiacols (10-30 g/ton pulp) and chlorocatechols/kinons (15-60 g/tonne pulp) are typically discharged from a bleached kraft pulp mill. From an ordinary size plant, this corresponds to 7-20 kg per day of chlorophenols and chloroguaiacols, and 10-40 kg per day of chlorocatechols/kinons (Statens naturvårdsverk 1987).

The total discharge of 4,5,6-trichloroguaiacol and 3,4,5-trichloroguaiacol from Swedish kraft pulp mills at the beginning of 1980s has been estimated at 22-24 tonnes/year (Neilson et al. 1989). The major part of this was emitted into the Baltic Sea, primarily into the northern Gulf of Bothnia. Measures taken in the process have led to a rapid decrease of the discharge and the yearly emitted amount of these compounds at the end of 1980s can be estimated at less than 2 tonnes (Neilson et al. 1989).

Levels in the environment

Chloroguiacols have been detected in sediments in the Bothnian Sea in a decreasing gradient up to 150 km from the coast (Kirkegaard & Renberg 1987). As pulp mill effluents are the only identified source of these compounds, the findings indicate a large scale dispersal of these discharges.

The exposure of biota to these compounds is verified from analysis of bile from perch caught in the pulp mill receiving waters. 3,4,5-trichloroguaiacol and 4,5,6-trichloroguaiacol concentrations between 100 and 100 000 ng/g wet weight were found (Söderström et al. 1988), which calculated as bioconcentration factor (BCF) for fish bile correspond to about 65 000.

Baltic salmon and brown trout collected in Finnish rivers were analyzed for ten different chloro-phenols with 3,4- dichlorocatechol as the main component in concentrations 19-90 ng/g wet weight (Paasivirta et al. 1985).

Chloro-veratrols and chloro-anisoles have been detected in old fiber deposits outside pulp mills and in invertebrates from long-term Baltic Sea littoral model ecosystem experiments (Neilson et al. 1989). Paasivirta et al. (1987) also report chloroveratroles in Baltic Sea salmon at concentrations of 2-3 ng/g fresh weight.

9.4.9 Dioxins

Dioxins are used as a collective name for polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Some of the total of 210 substances are extremely toxic. This, in combination with persistency and bioaccumulation properties, give rise to environmental concern.

In order to obtain a measure of the additive effect of the discrete substances, each congener has been given a factor based on its enzyme induction effect in relation to the most potent substance in this respect 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Hazard assessment of the unintentional release of dioxins is difficult, among other things, due to the extreme species variability in sensitivity.

Dioxins are found as contaminants in chemical products formed in processes involving chlorine, e.g., in pesticides (phenoxyacids) and wood preservatives (chlorophenols). They are also formed and emitted in waste incinerators and metal smelters, in accidental fires in capacitors containing PCB, etc. Dioxins released from different sources can, to a certain extent, be distinguished by the "fingerprint", i.e., the pattern of chromatographic peaks.

Environmental levels

The formation and release of dioxins from pulp mill bleacheries is illustrated by the study by Rappe et al. (1987). Sediment levels of chlorinated dibenzo-p-dioxin8 and dibenzofurans decreased in a pattern similar to that of guiacols from the pulp mill towards the open sea. The dominating congener in the sediment close to the plant was 2,3,7,8-TCDF. Dioxins in perch from the receiving waters *were* found in concentrations up to 19 pg/g fresh weight.

Dioxins, mainly particle bound, are transported by air and thus globally distributed. The emission and transport on a more narrow scale is illustrated by a study on dioxins in settling particulate matter in the Stockholm archipelago and offshore in the Åland Sea (Broman et al. 1988). The flux of dioxins by this mechanism indicated a rapid decline with increasing distance from the city. The estimated flux at the offshore station was 0.7 - 1.0 mg TCDD- equivalents/km^2/month. It could not be established in this study the contribution to this flux by air deposition as separated from a re-suspension mechanism, but the major part was considered to be a result of fallout. Also, the congener profile changed with increasing distance from the emission sources.

Dioxins are analyzed and found present in recent as well as in deep older sediments with 20 - 50 times higher concentrations in the recent layers. The toxic congeners of dioxins are present only in the recent sediment (Statens naturvårdsverk 1988).

Dioxin levels in herring homogenates from the Southern Baltic Proper, representing a time series from 1976 to 1985, vary between 3 and 5 pg TCDD-equivalents/g fish with a single extreme value of 14 pg/g in 1976. It is not possible to see any time trend from the limited number of analyses performed (Statens naturvårdsverk 1988).

Analysis of muscle homogenates of pike (*Esox lucius*) collected during the period 1969 - 1985 in two Swedish lakes, one in the north (Storvindeln) and one in the south (Bolmen) reveals no obvious time trend (Statens naturvårdsverk 1988). The TCDD- equivalents ranged from 1 to 4 pg/g in Bolmen and 0.3 - 1 pg/g in Storvindeln. The highest level in Bolmen was found in 1969 (Statens naturvårdsverk 1988).

Pooled samples of herring from the Swedish west coast contained lower levels of dioxins (1.8-3.4 pg TCDDeq/g wet weight) than samples from the Baltic Sea (6.7-9-0 pg/g) (Bergqvist et al. 1989a). The congeners present at the highest concentrations were 2,3,4,7,8-PeCDF and 2,3,7,8-TCDF. Levels of dioxins in cod were low and close to the limit of quantitation (0.1-0.2 pg/g).

Levels in relatively few wild salmon analysed from the Baltic Sea were generally 5-10 times higher than in herring (Rappe et al. 1989), most likely as a consequence of higher lipid levels in salmon. The levels in reared salmon were about the same as in herring. Vuorinen and Paasivirta (1988) found no dioxins (detection limit 2 pg/g wet weight) in salmon from the Gulf of Finland, while salmon from the Baltic Proper and from the Gulf of Bothnia contained significant amounts.

Three muscle homogenates of guillemot *(Vria aalge)* from Stora Karlsö collected in 1971, 1975 and 1979 show levels between 76, 17 and 17 pg TCDD equivalents/g respectively, while samples of blubber fat from grey seals contained 8 - 27 pg/g (Statens naturvdrdsverk 1988).

The dioxin levels in seals from different water areas around Scandinavia including Spitzbergen are surprisingly similar (Bignert et al. 1989). No significant age or sex differences in PCDD and PCDF levels could be established. The levels in seal blubber were of the same magnitude as the levels found in humans.

Analyses of human milk from 1972,. 1976, 1980 and 1984-85 (Norén 1987) indicate a decreased load on the Swedish population reflecting decreasing levels in the environment or possibly as a consequence of public information about environmental pollutants in certain foodstuff.

9.4.10 Balogenated organic matter

A variety of halogenated substances are formed and released by human activities. The introduction of halogen into an organic molecule usually render it a higher degree of stability and resistance to chemical and biological degradation. Halogenated man-made substances of various origin are today found widely distributed in the environment thus giving rise to special concern.

As a complement to the specific chemical analysis, development of analytical methods for TOCl (total organic bound chlorine), EOCl (extractable organic bound chlorine) and AOX (adsorbable organic halogenated matter) has made possible studies on the significance of the presence of synthetic halogenated material in the environment.

Special attention has in this respect been paid to the chlorine bleaching process in the wood pulp industry . Demonstration of specific chlorinated substances and e.g. AOX in water, sediment and biota outside pulp mills indicates a large-scale pollution of the Baltic Sea caused by effluents from the chlorine bleaching process (Södergren 1989).

The total discharge of chlorinated organic matter measured as AOX from the Swedish forest industry in 1988 was estimated at about 13 000 tonnes (Betge 1989). Most of this reaches the Baltic Sea and especially the Gulf of Bothnia.

Levels in water

Discharged effluents from bleached pulp mills have been traced in the receiving water up to 10-15 km from the outlet (Abrahamsson & Xie 1983, Xie et al. 1986, Jonsson et al. 1986). Since AOX also originates from other sources, it is not possible to assess the influenced area very precisely. Background levels, of hitherto unknown origin, increase from north to south in the Baltic Sea area (Enell and Wennberg 1988):

Bothnian Bay 10 - 15 µg/l
Bothnian Sea ca 20 µg/l
Baltic Proper and Kattegat 40 - 50 µg/l

Levels in sediment

Local and regional gradients in surficial sediment contamination by chlorine bleachery effluents have been established from EOCl analyses of samples taken outside pulp mills. Very high concentrations of EOCl, generally above 1 000 $\mu g/g$ IG (loss on ignition) are found in the sediment close to the mills . This is 50 - 250 times higher than in areas not exposed to local discharges (Håkansson et al. 1988).

The EOCl levels in the Baltic sediment in areas without point sources decrease from north to south (Södergren 1989). The levels in remote lakes and in estuaries with only municipal discharges without any particular local source of pollution were about 10 - 30 $\mu g/g$ IG compared to about 80 $\mu g/l$ in the central part of the Bothnian Sea and about 1800 $\mu g/g$ close to pulp mills.

Specific chlorinated phenolic substances (e.g. guaiacols) and dibenzo-p-dioxins/dibenzofurans were analyzed in sediment samples 150 km from the coast in the south central Bothnian Sea and in the northwestern Baltic Proper (Rappe et al. 1987, Kirkegaard & Renberg 1987). Dibenzo-p-dioxins/dibenzofurans also emanate from other sources. Chlorinated guaiacols however have never been shown to be discharged from sources other than pulp bleacheries. These observations have been taken as evidence of a large-scale pollution of the Baltic Sea from pulp mills.

Levels in biota

Discharged chlorinated substances are accumulated in organisms in the receiving water (Södergren 1989). The content of EOCl in muscle of perch, *Perca fluviatilis*, from the receiving water of a kraft pulp mill showed a decreasing gradient from the discharge point with the highest values of about 400 mg EOCl/kg fat weight. However, the generally high background level, even in inland lakes in remote areas in Sweden, of about 120 mg EOCl/kg in perch make an evaluation of the size of the influenced area rather uncertain. Also snails, *Lymnaea* sp., accumulate EOCl from bleached pulp mill discharges. The influenced area traced was of about the same size as that for perch.

At present, *only* about 5% of the total EOCl-load in fish from pulp mill receiving water has been identified (Kirkegaard & Renberg 1988).

Biological effects

Comprehensive ecotoxicological studies within the Swedish project Environment/Cellulose were carried out on fish and bottom fauna in the receiving water of a kraft pulp mill with chlorine bleachery (Södergren 1989). The results from the field studies were supplemented by laboratory experiments with whole effluents as well as with single chemical components identified in the effluents.

Detrimental effects of the discharge of bleached kraft mill effluent were documented on invertebrate and fish populations as well on individual organisms and organs.

Physiological disturbances were observed in perch at all stations studied with a successive decrease in response from the point discharge towards the sea. Still, disturbed reproduction and increased susceptibility to infections was indicated at a distance of 10 km from the discharge (Andersson et al. 1988a,b, Larsson et al. 1988).

Also pathological changes in fish in the form of fin erosion and skeletal deformation were associated with the discharge from the pulp mill (Bengtsson 1988a,b, Bengtsson et al. 1988).

The recruitment of fish was generally reduced in the receiving water of the pulp mill. Species with pelagic larvae such as herring were most sensitive to the effluent (Sandström & Thoresson 1988, Sandström et al. 1988).

Exposure of benthic crustaceans to effluents resulted in deformation of embryos and lower hatching frequency and survival (Sundelin 1988).

In summary, the results of the coordinated biological and chemical investigations on discharges of bleached kraft pulp mill effluents show a graded influence on the receiving waters which was characterized as:

local impact (O-10 km) - occurrence and levels of organochlorine residues in biota
regional impact (O-SO km) - zoophysiological and ecological effects
large-scale impact - organochlorine levels in the water and sediment.

9.4.11 Phthalates

Phthalates (phthalic acid esters) are widely used as a plasticizer in PVC plastics, in lubricants, in paints, etc. The annual world production is very large and phthalates are quantitatively one of the dominant synthetic chemical products in the world.

The chemical properties of phthalates are varying and depend on the characteristics of the chains - length etc. They are photochemically rather stable, but undergo hydrolysis to a certain extent and they tend to bioaccumulate.

The acute toxicity is low, but sublethal effects are reported at contaminant concentrations which are not unrealistically high.

Environmental levels

Emitted phthalates are mainly distributed to the air, governed by short-range and long-range transport (Thurén 1988)

Few analyses of- significance for the Baltic Sea environment have been reported. Sediment levels of di-2-ethylhexyl phthalate (DEHP) of 100 ng/g at the highest have been found, with corresponding values for dibutylphthalate (DBP) of 60 ng/g. Seal blubber contained 0.7 - 10.6 mg phthalates/kg extracted fat (Statens naturvårdsverk 1987).

9.4.12 Nonylphenol

Nonylphenol is an environmental contaminant present as a result of degradation of nonylphenol ethoxylates used as a large volume surfactant. The annual usage of nonylphenol ethoxylates in Western Germany was 18 500 tonnes (Huber 1985) and in Sweden 3 000 - 5 000 tonnes (Granmo et al. 1986). The discharge of nonylphenol from Swedish sewage treatment plants is calculated to be 100 - 1 000 tonnes per year.

Nonylphenol is persistent, bioaccumulating (Ekelund et a1.1989) and highly toxic to aquatic organisms (e.g., Granmo et al. 1989).

There is no available information about concentrations of nonylphenol in natural waters. This is a shortcoming as concentrations around 500 ug/litre in waste water from sewage treatment plants have been reported from a Swedish investigation (Granmo et al. 1986).

SUMMARY

Many anthropogenic organic compounds are found in the water, sediments and biota of the Baltic Sea. This load is the sum of several sources; riverine input, direct discharges, and airborne transport from sources mainly in the northern hemisphere.

In general, the residues of organic contaminants in fish from the open sea in the Baltic Area are higher than those from the central North Sea or the Atlantic. These differences are partly due to the differences in the receiving water volume and the rates of water exchange in the sea areas mentioned, but may also reflect different amounts of inputs from atmosphere, rivers and via direct discharges. The relatively low water temperature affecting the degradation rate might also be a factor of relevance.

Because of great analytical difficulties of measuring many of the organic pollutants in water, sediment and biota, and due to the limited knowledge of biological and ecological effects caused by persistent, bio-accumulating and toxic compounds, it is not possible to perform a hazard assessment of the present total load of organic contaminants on the Baltic Sea fauna and flora. Biological effects, however, seen as morphological deformation and reproductive impairment in seals, have been correlated to the load of organic pollutants, in the first place PCBs. Physiological effects, diseases and skeletal abnormalities are also verified in coastal fish populations in areas polluted by industrial discharges including organochlorine compounds.

It should also be stressed that changes of the biomass density in the sea exert influence on the levels of bioaccumulating substances recorded in biota. For a better evaluation of the causal factors of the distribution, levels and temporal trends of organic contaminants this factor and its variations over space and time must be considered in the future.

Emission controls applied for some well-known contaminants have reduced their levels in the environment. However, despite actions they are still present in amounts which are not acceptable.

There is a clear downward trend for DDT concentrations in the Baltic Sea from the 1970s to the 1980s; however, the environmental levels, as mirrored by comparable data from herring, are still higher than 'in the Skagerrak area. A small temporary increase in DDT concentrations in water and biota in the southern Baltic Proper from 1983 to 1985 was evidently a result of a temporary use of DDT for forest pest control in one country.

Also for polychlorinated biphenyls (PCBs) there was a clear downward trend in concentrations in biota from the 1970s to the 1980s and concentrations now seem to have stabilized at a lower level. The downward trend is most clearly visualised in the Baltic Proper. Comparable data on herring, however, indicate that the PCB levels are still higher in the Baltic than in the Skagerrak area.

Due to the ban on technical hexachlorocyclohexanes (HCHs), the Baltic Sea water concentrations of a-HCH, the main component of, have dropped considerably.

There is some concern about polychlorinated camphenes (PCCs) still in use as a pesticide. The slow rate of degradation and the bioaccumulative properties lead to PCC levels in the marine environment which are not possible to determine adequately due to the complex composition of the product. As a result, it is not possible to make a reasonable hazard assessment on the present use of PCCs.

There is an evenly distributed background contamination of petroleum hydrocarbons (PHC) in the open Baltic Sea, which seems mainly to be the result of atmospheric deposition of polyaromatic hydrocarbons (PAH). The concentration levels of single PAHs are in the range of ng/l or less. From the comparable data available at present (after 1982), however, it is not clear whether any trend can be established.

Contamination by saturated and aromatic petroleum hydrocarbons is higher in coastal waters and particularly close to urbanised areas, with a steep decline moving to open waters.

Adsorption to settling particular matter leads to high concentrations of PHC in sediments in accumulation areas under urban influence. The ecological effects of this chronic exposure on marine organisms are largely unknown.

Evaluation of analytical results obtained using the W-F method, which is only group specific, must be done carefully because of very large qualitative differences in PHC from different sources. The W-F method used for screening, and thus useful for large number of samples, must be supplemented by specific analyses on selected samples. Compound specific investigations are necessary in order to obtain reliable information on the input and fate of PHC in the Baltic Sea. To determine the specific PHC compounds in marine samples, measurement by gas chromatography/mass spectrometry (GC/MS) should be applied.

Data on contaminants other than DDT, PCBs, HCH and PHC are generally few and scattered and produced with different, non-standardised analytical methods. This implies that it is not possible to evaluate the distribution patterns or temporal trends in the environmental levels of these many contaminants.

Among the "new contaminants" there is an increasing list of identified compounds that could pose a threat to the environment according to their analogies with known pollutants. Examples of this are brominated biphenyls, brominated diphenylethers and chlorinated thiophenes.

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